Microhabitat influence on young-of-year fish assemblages within the vegetated beds of the St. Louis River estuary

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Abstract

Coastal wetlands within the Laurentian Great Lakes provide crucial habitat for an abundance and diversity of fishes during larval and juveniles stages; however, young-of-year fish habitat association studies at the microhabitat scale are rare. Because studies at this fine scale are crucial for understanding the habitat properties that support fishes at this critical life stage, I examined larval and juvenile fish assemblage structure in relation to microhabitat variables within the St. Louis River estuary, a drowned river mouth of Lake Superior. Fish were sampled in aquatic vegetated beds throughout the estuary, across a gradient of aquatic vegetation types and densities. Canonical correspondence analysis, relating species abundances to environmental variables revealed that principally microhabitat variables, including plant species richness, depth, and aquatic plant cover explained difference in fish assemblages. In addition, I evaluated the specific habitat associations of non-native Tubenose Goby (Proterorhinus semilunaris) to provide insight on habitat overlap with native species and further invasion risk. Tubenose Goby were nearly ubiquitous (75% if study sites) and abundant (0.59 fish/m^2) in the study site. Non-metric multidimensional scaling revealed that sunfish, Tadpole Madtom and Johnny Darter have the most potential for habitat overlap with Tubenose Goby. Generalized linear modelling revealed that Tubenose Goby occupied a specific microhabitat within vegetated beds consisting of dense cover and high plant species richness. Results from this microhabitat analysis at this critical life stage has potential to inform wetland management and restoration efforts within the St. Louis River estuary and other Great Lake Coastal Wetlands.

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Introduction

Coastal wetlands within the Laurentian Great Lakes provide crucial habitat for an abundance and diversity of fishes (Jude and Pappas 1992). Owing to their high productivity, diverse structural features and reduced wave energy, they serve as feeding areas and spawning grounds for adult fish and as important nurseries for larvae and juveniles (Hoffman *et al.* 2015; McDonald *et al.* 2014). Great Lakes coastal wetlands are also areas of concentrated human development which have resulted in dramatic declines in wetland acreage and condition. Over the past two centuries, 60-80% of wetlands in the Great Lakes region have been removed or degraded (Dahl 1990; Comer *et al.* 1995). Despite this loss, the remaining coastal wetlands are still areas of disproportionately high fisheries support (Trebitz and Hoffman 2015).

The successful restoration of degraded or removed wetlands depends on identifying the specific properties of remaining wetlands that support an abundance and diversity of fishes. These habitat functions are the ways in which a habitat contributes to the growth, survival or reproduction of a species, population or assemblage (Roundtree and Able 2007). Habitat, or a designated area that provides resources for a given species or assemblage, can be defined at many scales from whole bays or estuaries to patches of aquatic vegetation within a wetland (microhabitat). Research investigating habitat functions at all scales is necessary for identifying coastal wetland properties that support different fish assemblages.

Several studies have described differences in fish assemblages among Great Lakes habitats at various spatiotemporal scales and documented characteristics that influence species distribution and abundance (Bhagat and Ruetz 2011; Cvetkovic *et al.* 2010; Trebitz *et al.* 2009). It is well understood that coastal wetlands, due to warmer waters and higher productivity, support an abundance and diversity of prey items (benthic macroinvertebrates and zooplankton) that in turn support many resident forage fish such darters, minnows and suckers as well as larval and juvenile fish (Jude and Pappas 1992). Woody debris, gravel sediments and aquatic vegetation (emergent and submerged) provide ample spawning substrate for both resident and transient species (Judas and Pappas 1992). Additionally, coastal wetlands have been identified as especially important nursery habitat for young fish (Brazner 1997; Chubb and Liston 1986; Hoffman *et al.* 2015; McDonald *et al.* 2014; Petering and Johnson 1991) meaning that they produce a disproportionate number of individuals that recruit to the adult population relative to other habitats (Beck *et al.* 2001). Despite this importance, few studies have investigated the relative roles of multiple scale influences on larval and juveniles fish within coastal wetlands (Hook *et al.* 2001; Tanner *et al.* 2004; Parker *et al.* 2009). This information is critical for understanding the functional role of coastal wetlands for fishes during early life stages to better manage and restore coastal wetlands that support Great Lakes fishes.

The following study investigated environmental influences on larval and juvenile fish assemblages at the microhabitat scale within a Great Lakes coastal wetland

and had four main objectives. The first objective (Chapter 1) was to describe differences in larval and juvenile assemblages among microhabitats within aquatic vegetation beds, and subsequently identify environmental variables associated with the distribution and abundance of larval and juvenile fish. The second objective (Chapter 1) was to identify the predominant habitat associations of three wetland-dependent species of interest (sunfish, *Lepomis* spp.; Rock Bass, *Ambloplites rupestris*; and Black Crappie, *Pomoxis nigromaculatus*) and describe factors indicative of quality nursery habitat as measured by fish species richness. The third objective (Chapter 2) was to document the distribution and abundance of young-of-year Tubenose Goby, a non-native species of concern, within aquatic vegetation beds to assess their microhabitat associations. The fourth objective (Chapter 2) was to evaluate potential microhabitat overlap between Tubenose Goby and native species that also use coastal wetlands.

Chapter 1: Distribution and abundance of young-of-year fishes in a Lake Superior coastal wetland; evaluating microhabitat associations within aquatic vegetation beds

Introduction

Great Lakes coastal wetlands provide habitat for a diverse fish assemblage; most of these species are captured in coastal wetlands as larvae and juveniles, demonstrating their important role as spawning and nursery habitat (Jude and Pappas 1992), and Great Lakes fisheries support (Sierszen *et al.* 2012; Trebitz and Hoffman 2015). Relative to nearshore habitat throughout the Laurentian Great Lakes, coastal wetlands provide young-of-year (YOY) fish warm water habitat with abundant benthic macroinvertebrates and zooplankton, which are essential for rapid growth (Shoup *et al.* 2007; Cooper *et al.* 2012). There is widespread agreement that coastal wetlands provide vital and diverse habitats for young fish (Chubb and Liston 1986; Petering and Johnson 1991; Hook *et al.* 2001; Tanner *et al.* 2004; McDonald *et al.* 2014; Hoffman *et al.* 2015). Assemblages of YOY fishes vary among wetlands based on both abiotic (physical and chemical; Miehls and Dettmers 2011; Parker *et al.* 2012) and biotic factors (French 1988; Welker *et al.* 1994). Whereas climate and lake morphology can affect distribution across broad landscapes, local variation in temperature, dissolved oxygen, turbidity and pH can act to structure assemblages within that landscape (Jackson and Harvey 1989) and even within patches of aquatic vegetation (Jacobus and Ivan 2005). Biotic factors, such as habitat structure, prey availability and competition also influence YOY fish assemblages at the

microhabitat (i.e., patch) scale (Rountree and Able 2007; Klecka and Boukal 2014).

Among wetland habitat characteristics, the presence and type of submerged aquatic vegetation (SAV) has a strong influence on YOY fish assemblages in Great Lakes coastal wetlands (Chubb and Liston 1986; Tanner *et al.* 2004; Cooper *et al.* 2012). SAV provides spawning substrate for many adult fishes (Lane *et al.* 1996), which greatly influences larval distribution at early stages (Tanner *et al.* 2004). Along with emergent and floating leaf plants, SAV regulates temperature (Carter *et al.* 1991), and retains sediment and nutrients (Krieger 2003), which improves overall water quality (Sierszen *et al.* 2012). Additionally, organic matter derived from SAV contributes to larval fish production in coastal wetlands (Hoffman *et al.* 2015). By providing substrate for epiphytic algae, SAV can increase primary productivity and thereby prey biomass for larval fish (Dibble *et al.* 1996). Finally, SAV influences predator-prey interactions by acting as cover from larger predators (Crowder and Cooper 1982; Trebitz *et al.* 1997).

SAV can supersede large-scale abiotic factors and local water quality as the most important factor influencing fish assemblages (Uzarski *et al.* 2005; Cvetkovic et al 2010; Hanson *et al.* 2015; Janetski and Ruetz 2015). In coastal wetlands, vegetated habitats have higher fish species diversity and YOY fish abundance than non-vegetated habitats (Chubb and Liston 1986; Leslie and Timmons 1994; Trebitz *et al.* 2009). However, how species vary across a gradient of SAV and aquatic vegetation densities and complexity within coastal

wetlands is not well-understood (Petering and Johnson 1991; Cooper *et al.* 2012). Moreover, identifying characteristics of vegetated habitats associated with specific YOY fishes is needed to inform current and future coastal wetland habitat restoration efforts that aim to create fish habitat (GLRI 2010).

Densely vegetated habitats are difficult to sample. Light traps are easy to deploy and retrieve; however, they cannot be used to determine fish densities, limiting the type of analyses available to determine habitat associations. Turbidity and light source affect the quantity and species composition of larvae in samples (Matchetti *et al.* 2004). Quantifying aquatic vegetation density is also challenging and vegetation metrics vary widely. Many researchers employ visual estimates of percent aquatic vegetation cover or biovolume (Bhagat and Ruetz 2011; McDonald *et al.* 2014); however, these estimates can vary substantially among observers (Klimes 2003) and are biased when compared to measured values (Killourhy *et al.* 2016). Often aquatic vegetation cover estimates are categorized (i.e. low, medium, and high) before analysis (Tanner *et al.* 2004; Randall *et al.* 1996) which likely dampens bias and inaccuracy. But, while categorical analysis may reveal broad fish associations and is well suited to across wetland or macrohabitat (~1 km) studies, fine scale continuous quantification is needed to fully describe YOY habitat associations. Aquatic vegetation quantification is often paired with floristic surveys to describe fish associations with plant species composition (Cvetkovic *et al.* 2010), form and complexity (Hook *et al.* 2001). Detailed floristic surveys require extensive time and labor and are therefore rare.

However, they can reveal meaningful associations between fish assemblages and the plant community, and offer valuable insights into the functional role of aquatic vegetation (Cvetokovic *et al.* 2011).

Because fine-scale early life history studies in coastal wetlands are rare, and YOY habitat associations need identified to inform restoration efforts, I had two objectives in this study. My first objective was to describe differences in larval and juvenile assemblages among microhabitats within aquatic vegetation beds in Great Lakes coastal wetlands, and subsequently identify both abiotic (water chemistry) and biotic (productivity, plant composition, and ecological condition) variables associated with the distribution and abundance of larval and juvenile fish. My second objective was to identify the predominant habitat associations of three wetland-dependent species of interest (sunfish, *Lepomis* spp.; Rock Bass, *Ambloplites rupestris*; and Black Crappie, *Pomoxis nigromaculatus*). These three species are found throughout the Great Lakes basin (Judas and Pappas 1992) and yet their microhabitat associations within coastal wetlands as YOY are still poorly understood. In addition, I describe factors indicative of quality nursery habitat as measured by fish species richness.

Within the St. Louis River estuary (hereafter referred to as SLRE), the largest drowned river mouth coastal wetland in the Great Lakes, I sampled YOY fish across a continuum of both abiotic and biotic conditions, including aquatic vegetation cover, composition and form (i.e. floating leaf, emergent, and SAV). Using fine-scale habitat characterization at the aquatic vegetation-patch scale

(i.e., microhabitat scale) to reveal fish assemblage and species-specific habitat associations, this study provides basic knowledge pertaining to the early life history requirements of coastal wetland fishes and information to guide ongoing Great Lakes coastal wetland restoration efforts.

Methods

Study location

The St. Louis River is the second largest tributary to Lake Superior, with a watershed of 9,280 km^2. At its terminus, it forms a 49 km^2 drowned river mouth between Duluth, MN and Superior, WI (Fig. 1). The middle and upper portions of the SLRE are relatively shallow (<3 m) with numerous bays, tributary mouths and islands, creating a mosaic of emergent wetlands and SAV beds. In contrast, the lower SLRE includes an industrial harbor (the largest commercial port in the Great Lakes by cargo tonnage) bordered by urban development with few vegetated areas. On the system's eastern end, a barrier beach encloses Allouez Bay; the relative protection afforded by the barrier beach fosters large emergent aquatic vegetation and SAV beds. The local watersheds of Allouez and Pokegama bays deliver high clay sediment loads, causing high turbidity relative to the rest of the SLRE.

The aquatic plant community within the SLRE is dominated by water celery (*Vallisneria americana)*, but is comprised of at least 330 species (Danz *et al.* 2017). This plant species diversity is complemented by numerous resident and

migratory fish species (at least 60) that use the estuary (J. Hoffman, unpublished data).

Site locations

The SLRE has an upriver-downriver gradient in both water chemistry and fish species composition (Peterson *et al.* 2011, Bellinger *et al.* 2016). I selected twelve vegetated sites spread across the SLRE to represent the range of conditions present (Fig. 1). Based on inspection of aerial imagery, the SAV model developed by Angradi *et al.* (2013), previous floristic data (Danz *et al.* 2017) and field reconnaissance, sites were selected to represent the density and diversity of aquatic vegetation types available. At each site, three microhabitats were selected for sampling based on a visual assessment of the abundance of different aquatic plant forms. Microhabitat categories included only SAV (n=13), dense floating leaf plants (n=13), and emergent stands (n=10). Those only SAV sites were further divided into microhabitats based on percent areal cover (estimated visually): low (<50%, n=2), medium (51-70%, n=5), and high (>70%, n=5). A single transect (25 m) was laid out within each microhabitat for fish and aquatic vegetation sampling.

Water quality measures and zooplankton collection

From 26 July through 3 August, 2016, temperature, dissolved oxygen, specific conductivity, pH and turbidity were measured *in situ* at the start and end points of each transect, 0.5 m below the surface (Hydrolab HL4 multiparameter sonde,

OTT Hydromet, Loveland, CO). Also, two subsurface water samples (0.5 m depth; 1L) from the start and end of each transect were composited for chlorophyll-a analysis. Samples were placed in a cooler for transport and vacuum filtered (47 mm GF/F) in the laboratory within 9 hours of collection. Chlorophyll-a concentrations were determined using fluorescence (Trilogy® Laboratory Fluorometer, Turner Designs, San Jose, CA) after a 24-hour 90% acetone extraction (EPA Method 445.0).

I collected a vertically-integrated sample (0.25 m off bottom to surface) of zooplankton using a diaphragm pump (Jabsco 3460). Water from each end of the transect (40 L) was composited and sieved (500 µm). Zooplankton were preserved in 95% ethanol. In the laboratory, samples were further sieved (1000 µm) to remove algae because it prevented efficient sorting. Both portions of five zooplankton samples (500-1000µm and >1000µm) were sorted by order and compared to ensure that sieving was not selecting specific taxa; within both size categories, cladocerans were dominant (~90% by abundance). The >1000µm samples were then filtered on to pre-weighed 500µm mesh filters, dried for 24 hours at 60°C, and weighed (AX2015 Delta Range balance; ±1 mg).

Aquatic Vegetation

In early August, I conducted a hydroacoustic survey of each site using a down-looking, single beam transducer (Biosonics DT-X digital echosounder; 6° beam, 420 kHz). Data were collected using Visual Acquisition software (version 6.2, Biosonics Inc., Seattle), transmitting 5 pings per second to obtain, on average,

4.5 pings per meter (linear distance). A 10 x 10 m coordinate grid was created in Arcmap and loaded onto an onboard GPS unit. This grid was used to establish hydroacoustic survey transects that were10 m apart and positioned perpendicular to fish sampling transects when possible.

In addition, between 30 August and 7 September 2016, I surveyed aquatic vegetation at each transect, using a stratified random design to assign sample locations. At each stratum: beginning, center, end of each transect, a randomly selected grid point (developed for the hydroacoustic survey) was selected for sampling. I placed a 1 x 1 m quadrat at each selected grid point, then assigned a total percent cover estimate for the area within the quadrat. All plants were then removed (by hand or with a rake), identified to species, and assigned a percent cover value by species (MPCA 2014). Any additional species surrounding the boat, but not within the quadrat, were also noted and were used in plant species richness calculations. Any plants unidentified in the field were collected, kept cool, and transported back to the laboratory for identification.

Plant survey data from quadrat sampling was used to calculate plant species richness and abundance-weighted Coefficients of Conservatism (wC). The wC metric is an aquatic vegetation-based assessment of overall wetland condition. It is based on the abundance and numerical rating (C-scores) of each individual species within a site. The C-values represent an individual species' habitat fidelity and disturbance tolerance. I used regionally assigned C-values for Minnesota (MPCA 2014) for my calculation. The estimated species cover from

each survey was also used to calculate the relative cover of plant forms (floating leaf, emergent, and SAV) for each microhabitat.

I used Visual Habitat software (version 2.0, Biosonics Inc., Seattle, WA) to analyze the hydroacoustic data, calculating summary statistics for groupings of ten consecutive pings. For georeferencing, I assigned the latitude and longitude associated with each group's center. I exported the georeferenced summaries to ArcMap 10.4.1 (ESRI, Redland, CA) and calculated average aquatic vegetation cover (percent pings classified as plant), average depth, and total biovolume for each transect, only including ping groupings within 25 m of the transect. To index aquatic vegetation patchiness, the average aquatic vegetation cover coefficient of variation was also calculated.

Fish collection

YOY fish were collected using a benthic sled (1 m x 1 m net opening, length 3 m, 500 µm mesh net) that was pulled along a 25m transect within each microhabitat. Fish were only collected during daylight hours to insure safe and proper deployment. Only YOY fish were retained; older fish were identified and released. Material in the net, including vegetation and debris, was triple rinsed in river water, sieved (500 µm), and preserved in 95% ethanol. In the laboratory, each fish specimen was measured (total length [TL], ± 1 mm). All specimens were identified to species (Auer 1982), except *Lepomis* spp (sunfish). Bluegill (*Lepomis macrohirus)* and Pumpkinseed (*Lepomis gibbosus)* can hybridize and

are difficult to distinguish as larvae. Ten percent of fish from each size class were independently confirmed by a second taxonomist; no discrepancies were found.

Fish assemblage-habitat analysis

I used canonical correspondence analysis (CCA) to explore whether differences in the composition of YOY fish assemblages within vegetated beds were related to environmental variables. CCA is multivariate technique that offers a direct gradient approach to relate two multivariate datasets concurrently (Ter Braak 1986), typically species community composition and associated environmental variables. I used CCA to simultaneously identify fish species compositions in my study area and relate those compositions to the environment, including abiotic parameters that are known to vary across the estuary (i.e., turbidity, specific conductivity) and microhabitat factors that exhibit small scale variances (i.e., zooplankton concentration, aquatic vegetation cover, plant species richness). Volumetric species-specific densities for larvae and juveniles combined were calculated for each transect. Only species with >1% relative abundance and >5% site occurrence were included in the CCA to avoid undue influence from rarely encountered species. Because an overabundance of explanatory variables can limit the “constrained” nature of CCA, variables were excluded if redundant (e.g. aquatic vegetation cover and SAV cover) or highly-related (e.g. pH and dissolved oxygen) based on Pearson’s correlations ($r>0.8$). I used *cca* function in the *vegan* package (Oksanen *et al.* 2016) in R statistical software (v3.3.2, R Core Team) which is based on the algorithm detailed in Legendre and Legendre

(2012). Analysis of variance (ANOVA) residual permutation tests on the full model and individual model axes were performed using the function *anova.cca* to test for significance (Legendre *et al.* 2011). Both within-site and across-site coefficients of variation (CV) were calculated and compared to assist CCA interpretations.

In addition to CCA, I modelled fish species richness in a series of univariate generalized linear models (GLMs) to identify predominant models associating environmental variables with fish species composition. In contrast to linear models, GLMs fit data on a transformed scale while calculating variance on the original scale of the predictor. This allows for non-normal error distributions, which were common in my dataset. I fit 15 candidate fish species richness models, each with only one independent variable. I used maximum likelihood to estimate model variance parameters (θ) and coefficients. Each model was fit using a log-link function. I calculated Akaike Information Criterion values for small samples sizes (AICc). The AICc values estimate the quality of a model based on both goodness of fit and complexity. I compared each model's AICc to a null model's (intercept-only) AICc; more parsimonious models (lower AICc) indicate variable associations with fish species richness. The best models (lowest AICc) specify the environmental variables that had the strongest association with fish species richness.

Species-habitat analysis

I used the same 15 independent variables and univariate GLM approach to determine whether any environmental variables were significantly related to the distribution and abundance of Black Crappie, Rock Bass, and sunfish (*Lepomis* spp.). I fit negative binomial GLMs using maximum likelihood and a log-link function for each of my fish species. Negative binomial GLMs are commonly recognized as successful models for count data, particularly when the error distribution is non-normal and overdispersed when compared to Poisson models (Vaudor *et al.* 2011). AICc values were calculated and compared as previously described. All modelling was performed in R (R 3.3.2 statistical software, R Core Team).

Gear Biases Analysis

Because the benthic sled is an uncommon gear for capturing YOY fish in vegetated habitats, I evaluated two areas of potential bias: towing speed and size selectivity. For the former, I tested whether there was a significant correlation between tow duration (sec) and total fish abundance. For the latter, I constructed catch curves and tested whether there was a significant correlation between mean fish lengths (all species combined) and average transect depth, as well as mean fish lengths and average aquatic vegetation cover.

Results

Water quality measures and zooplankton abundance

Temperature (range: 21.6º - 28.4º C), dissolved oxygen (range: 35 - 100%), and pH (range: 7.23 - 8.14) varied little (CV < 60%), both across and within sites; and displayed relatively normal distributions (Fig. 2). High specific conductivity (>154 µS/cm; outliers in Fig. 2), was measured at the two most upstream sites. The Allouez Bay sites had substantially higher turbidity (> 40 NTU) than the other sites, resulting in moderate variability across sites (range: 0.6 - 55.4; CV 78%). Zooplankton concentration (range: 0.001 - 0.08 mg/L) variability was similar among (CV = 78%) and within sites (CV = 82%).

Aquatic Vegetation

I identified 29 aquatic plant species (Table 1). *Valisneria americana* (C-score of 6) occurred within most quadrats sampled (85%) and was the most abundant species encountered. Calculated wC scores had a small range and low among-site variation (range 4.11 – 6.45, CV 4% Table 2). Plant species richness however, had much higher among site and within site variation (52%, 47% Table 2) with the highest richness (14) at Munger Landing, a middle estuary site. Aquatic vegetation cover derived from my hydroacoustic analysis ranged from 8 to 98% with a normal distribution, confirming that my microhabitat designations based on visual estimates appropriately covered a gradient of aquatic vegetation cover. Patchiness had a median value of 42.7% with high values occurring in Allouez Bay (range 90 - 186%) and Pokegama Bay (range 83 - 98%), two high-turbidity sites. High patchiness also occurred in microhabitats with low aquatic vegetation cover (Pearson's r=-0.75). Aquatic vegetation form (floating leaf,

emergent, SAV) had moderate to high variation (Table 2) relative to water quality parameters and other aquatic vegetation metrics.

Fish collection

I captured a total of 1,135 larval and juvenile fish belonging to 12 taxa. The density of all taxa combined was 1.26 fish/m^3 (Table 3); densities ranged from 0.04 – 6.44 fish/m^3 among transects with the lowest densities in Allouez Bay and the lower estuary (0.63 and 0.16 fish/m^3, respectively) and the highest density in Radio Tower Bay (mean density=3.5 fish/m^3), the most upstream site. Tubenose Goby (*Proterorhinus semilunaris*) was the most abundant (46.4% of the total catch) and frequently encountered (75% occurrence) species. The combined abundance of the four most prevalent taxa; Tubenose Goby, sunfish, Black Crappie, and Rock Bass comprised 89.5% of the total abundance.

Fish community-habitat relationships

Three species - Smallmouth Bass (*Micropterus dolomieu*), Round Goby (*Neogobius melanostomus*), and Logperch (*Percina caprodes*) - were classified as rare by my criteria and were not included in the CCA. Common Carp (*Cyprinus carpio*) was also removed because of its skewed distribution (20 of 26 individuals were captured at the same micorhabitat). I included 11 environmental variables after removing pH and aquatic vegetation biovolume because they were highly correlated (coefficient >0.8) with dissolved oxygen and aquatic vegetation cover, respectively.

The full CCA model was highly significant ($p = 0.008$) and explained 51.5% of the variation (inertia) in the dataset. Of the seven axes produced, the first four axes were significant based on individual axis residual permutation tests ($p < 0.05$). The first two axes' canonical loadings were used to identify primary relationships among environmental variables (Fig. 3). Axis one revealed positive associations among aquatic vegetation cover, plant species richness, and zooplankton density; all were negatively associated with depth, patchiness, and turbidity. On axis two, turbidity was positively correlated with patchiness; both were negatively correlated with temperature. Axes three showed positive relationships among specific conductivity, zooplankton abundance, and aquatic vegetation cover, which were all negatively associated with depth and temperature. Axis four, confirmed the positive relationship between specific conductivity and zooplankton.

Of the explained variance in my CCA (51.5%), the first axis described 37.9% of the species-habitat relationship and was largely comprised of aquatic vegetation cover, depth and plant richness. This indicates that microhabitat factors contribute the most to the CCA's explained variance and therefore are critical components explaining differences in YOY fish assemblages. Tubenose Goby was the only species with a strong, positive correlation with the first axis (defined by species score $>|0.5|$), indicating that it dominated assemblages in shallow habitats with high plant species richness and dense aquatic vegetation. Conversely, several taxa, including Rock Bass, sunfish, Black Crappie, and

Spottail Shiner had strong, negative correlations with the first axis indicating a mixed fish assemblage in deeper, less densely-vegetated habitats.

The second axis, which accounted for 28.0% of the species-environment relationship (or CCA explained variance), was primarily composed of turbidity, patchiness, and temperature. Johnny Darter and sunfish had strong, positive correlations with axis two (i.e., associations with warmer, less turbid habitats), whereas Spottail Shiner and Black Crappie had strong, negative correlations with axis two (i.e., associations with cooler, more turbid habitats). Rock Bass, Spottail Shiner and Golden Shiner were positively correlated with axis three, indicating a positive association with specific conductivity. Golden Shiner was highly correlated with axis four confirming an association with specific conductivity.

Overall, the CCA revealed distinct larval and juvenile fish assemblages within vegetated microhabitats. Plant species richness, aquatic vegetation cover, and depth explained the most variability in my CCA, defining a gradient between two assemblages, one dominated by Tubenose Gobies in shallow, densely vegetated habitats, and one of primarily *Centrarchids (*sunfish, Rock Bass and Black Crappie) in deeper, more moderately vegetated habitats. Despite among-site differences from the upper to lower estuary in turbidity, temperature and specific conductivity, my results suggest microhabitat factors are more important than large-scale abiotic factors in structuring YOY fish assemblages.

Among microhabitats, fish species richness ranged from 1 to 7 (mean=3.7). Based on the univariate GLMs, several variables explained differences in fish

richness (based on a model fit better than the null model, Table 4). Among all variables included on my analyses, turbidity ranked as the best variable (lowest AICc) and was negatively correlated with fish species richness. In addition, species richness was positively correlated with plant species richness, floating leaf plant cover, SAV cover, chlorophyll-a and zooplankton concentrations.

Species-habitat relationships

Rock Bass was negatively associated with turbidity and aquatic vegetation patchiness and positively associated with specific conductivity. Turbidity was the most important variable influencing Rock Bass abundance based on AICc ranked performance of GLMs (Table 4). Like Rock Bass and overall species richness, sunfish were also negatively associated with turbidity; however, they had the strongest response to temperature and were associated with warmer habitats. In addition, sunfish were positively correlated with depth, floating leaf plant cover, and SAV cover. Only two Black Crappie models performed slightly better than the null model, turbidity and wC score. Poor model performance indicates a low association with any one variable; however, the positive association with turbidity is consistent with CCA results.

Gear biases

I did not find an operational or size selectivity bias. Total fish abundance was not significantly correlated with benthic sled tow duration ($p=0.42$, Pearson's $r=0.14$ Fig. 4a), and mean fish length was not significantly correlated with either average

transect depth (p=0.23, Pearson's r=-0.21; Fig. 4b) or average aquatic vegetation cover (p=0.78, Pearson's r= -0.048; Fig. 4c).

Discussion

I found that environmental factors operating at the microhabitat scale influence the distribution and abundance of YOY within Great Lakes coastal wetland aquatic vegetation beds. Despite large scale gradients in water quality variables (i.e. turbidity, specific conductivity, and temperature) across the SLRE, the CCA identified a combination of depth, aquatic vegetation cover, and plant species richness as the underlying explanatory habitat gradient associated with YOY assemblage differences. Although the final CCA explained only 51.5% of variability in the dataset, small scale differences in physical and biotic qualities of habitat explained more variance in YOY assemblage structure than did larger scale differences in water quality. In conjunction, GLM results established that high fish species richness was associated with aquatic vegetation characteristics, including composition and cover. However, wC score, my floristic measure of habitat quality, performed poorly. Both the CCA and GLMs indicated that YOY fishes responded directly to measures of aquatic vegetation cover and composition, but did not vary with respect to the wC score. Thus, when used alone, wC was a poor predictor of nursery habitat quality at the microhabitat scale.

Complementary to other studies in Great Lakes coastal wetlands, I identified aquatic vegetation as a critical factor associated with larval and juvenile fish distribution and abundance. While previous research has described habitat-associations generally among aquatic vegetation cover categories or presence-absence, this study revealed detailed habitat associations along a quantified cover gradient. For example, previous studies have indicated that juvenile Tubenose Goby prefer vegetated over non-vegetated habitats (Jude *et al.* 1992; Dopazo *et al.* 2008; Kocovsky *et al.* 2011). I further elucidated their stronger association with aquatic vegetation cover compared to all other species captured in this study, and showed their assemblage dominance in the shallowest, most densely vegetated and plant species rich microhabitats. Similarly, I documented larval sunfish abundance within aquatic vegetation concordant with previous research in both Lake Erie (Petering and Johnson 1991) and Lake Michigan (Chubb and Liston 1986) wetlands. I further identified their affiliation with floating leaf plant cover. This association with a particular aquatic vegetation form was distinct from other wetland-dependent Centrarchids, which responded primarily to water quality (Table 4).

In addition to aquatic vegetation cover, this study identified aquatic plant composition as a primary factor affecting both YOY fish assemblage structure and overall fish species richness. Where aquatic vegetation cover and plant richness were high, fish species richness was also high. This trend is evident at larger scales for adult fishes within and among Great Lakes coastal wetlands

(Randall *et al.* 1996; Brazner and Beals 1997; Cvetkovic *et al.* 2010) and large inland lakes (Weaver *et al.* 1997; Pratt and Smokorowski, 2003). The species composition and abundance of whole coastal wetland aquatic plant communities are influenced by water quality, varying with turbidity and nutrients (Lougheed *et al.* 2001; Trebitz *et al.* 2007; Croft and Chow-Fraser 2007), and physical and geological factors (Keough *et al.* 1999; Lougheed *et al.* 2001; Trebitz *et al.* 2011). At a finer scale within aquatic vegetation beds, previous studies demonstrate that community composition and abundance varies from the open water edge towards inner marshes along a depth gradient (Cardinale *et al.* 1998; Cooper *et al.* 2012). In this study, plant species richness had similar variances among and within sites (Table 2) indicating that YOY fishes respond to a microhabitat plant community and density gradient, in addition to the larger scale plant community differences among vegetated beds.

While this study did not explicitly test the functional role of aquatic vegetation as fish nursery habitat, previous research indicates that microhabitats with high plant density and diversity support high macroinvertebrate (Cyr and Downing 1988) and zooplankton abundances (Lougheed and Chow-Fraser 1997; Copper *et al.* 2012; Choi *et al.* 2014). So, in addition to predator protection (Harris *et al.* 2004), aquatic vegetation offers high prey abundance which fosters rapid larval growth (Welker *et al.* 1994). I also found a positive relationship between aquatic vegetation cover, plant species richness, and zooplankton concentration. Additionally, zooplankton concentration was a primary factor explaining fish

species richness, indicating that aquatic vegetation serves many functions for YOY fish.

Notably, high microhabitat wC scores were not associated with fish species richness variability and did not substantially influence YOY assemblages. Floristic indices are commonly used to assess biologic integrity, habitat quality, and restoration success (Nichols 1999; Croft and Chow-Fraser 2007; MPCA 2014); however, I did not find that wC scores were associated with particular YOY fishes or overall fish species richness within a coastal wetland system. This was caused by low variability across the estuary and within individual sites (Table 2). With a C score of 6, water celery's plant community dominance and ubiquity within the SLRE contributed to low variability among sites and an assessment of "fair" for all habitats (Table 3). Based on this finding, I recommend that researchers focus efforts on developing larval and juvenile fish-specific indices of habitat quality so managers do not have to rely on indices designed to assess general wetland ecological integrity or water quality. The GLMs indicate that aquatic vegetation, productivity (including both chlorophyll-a and zooplankton concentration), and turbidity are strongly associated with habitat species richness and are good candidates for assessing nursery habitat.

Secondary to aquatic vegetation characteristics, I found turbidity and temperature influenced YOY assemblages. Turbidity is a prominent abiotic factor influencing fish assemblages within drowned river mouths (Bhagat and Ruetz 2011; Janetski and Ruetz 2015) and coastal wetland systems (Brazner and Beals 1997; Trebitz

et al. 2009). High turbidity, caused by any combination of nutrient loading, sediment loading, and wind or wave action (Keough *et al.* 1999), directly impacts larval foraging success and predator-prey dynamics (Miner and Stein 1992; Shoup and Wahl 2009). Further, it can indirectly influence fish assemblages by limiting SAV growth and shifting plant species compositions (Lougheed *et al.* 2001). Within the SLRE, Angradi *et al.* (2013) found that high turbidity decreases the optimal growing depth for water celery congruent with my finding that turbidity was positively correlated with aquatic vegetation patchiness (a measure of cover variance) and negatively correlated with aquatic vegetation cover and plant species richness.

Turbidity tolerances vary among wetland-dependent species (Trebitz *et al.* 2007). Thus, it was not surprising that turbidity had a strong influence on the SLRE assemblages, consistent with other larval fish assemblage studies (Petering and Johnson 1991; Hook *et al.* 2001). This study identified a distinct assemblage dominated by Spottail Shiner and Black Crappie in highly turbid habitats reflecting known adult tolerances (Trebitz et a. 2007). Turbidity was a common factor influencing the distribution and abundance of all three individually modelled species (Table 4). Additionally, it had the lowest AICc of all fish species richness models, indicating that water clarity is an important attribute of nursery habitat.

Temperature's importance in this study's CCA was surprising given its short range and low variability both among and within sites (CV=0.07%, 0.02% respectively). While temperature often varies distinguishably over brief time

periods within wetlands (Stephenson 1990), my point measures of temperature were not adequate to characterize temperature at the scales to which fish respond (Baltz *et al.* 1987; Paradis *et al.* 2014). These measures likely reflect a system-wide temperature gradient previously described in the SLRE (Angradi *et al.* 2013; Hoffman *et al.* 2015) with warm waters in the upper estuary and sheltered bays and cool temperatures in the lower estuary, which is influenced by the Lake Superior seiche (Trebitz 2006). I confirmed that my point measures reflected this gradient by comparing them to mean daily averages calculated from continuous temperature data collected at nine sites across the estuary in 2016 (Peterson 2017, Environmental Protection Agency, preliminary data). Previous research also indicates that adult assemblages vary along the same lower to upper estuary axis as temperature (Peterson *et al.* 2011); thus, I believe the temperature-assemblage relationship depicted in the CCA reflects this estuary-scale association and possible adult spawning habitat preferences. Because several variables also vary along this axis from upper to lower river (*i.e.,* human impact, current, wave action, *etc.*) it is difficult to discriminate if fish are responding directly to temperature.

Our study identified microhabitat factors, specifically depth, aquatic vegetation cover, and plant species richness as more important than water quality, despite larger scale gradients in water quality across the SLRE. It is largely recognized that fish community compositions and environmental factors vary distinguishably at different scales (Jackson and Harvey 1989; Jackson *et al.* 2001; Magurran *et*

al. 2011), and that both local habitat features and regional factors influence species distribution and abundance (Brazner and Beals 1995; Brazner 1997; Hook *et al.* 2001; Uzarski *et al.* 2005; Trebitz *et al.* 2009). Few studies have investigated the relative roles of multiple scale influences on larval and juvenile fish within coastal wetlands (Hook *et al.* 2001; Tanner *et al.* 2004; Parker *et al.* 2009). Although my study had a limited temporal scale to address all regional factors that may influence fish assemblages, I provide evidence that late larval-early juvenile fish assemblages respond primarily to microhabitat variability. This corroborates Hook et al.'s (2001) conclusion that local habitat variables, specifically aquatic vegetation form and density (calculated as an index of habitat complexity), were more important than bay-wide factors (i.e. morphology and degree of human impact) in Les Cheneaux (Lake Huron) wetlands. Also, congruent with Hook *et al.* (2001) and Tanner *et al.* (2004), I detailed how species vary in their response to microhabitat factors. Black Crappie responded little to microhabitat differences but strongly to turbidity, indicating a system-wide response perhaps indicative of adult associations. In contrast, Tubenose Goby was evenly distributed across the estuary and responded strongly to microhabitat differences in aquatic vegetation cover and plant species richness.

In this study, I prioritized my sampling effort spatially rather than temporally. I am aware that this single sampling event, in late summer, does not provide me with any ability to describe likely seasonal assemblage shifts (Jacobus and Ivan 2005). For example, Yellow Perch were noticeably absent from my catch data,

likely because I sampled after their littoral to pelagic migration, which occurs after yolk-sac absorption for a period of 4-8 weeks (Post and McQueen 1988; Dettmers *et al.* 2005). However, yolk-sac and early stage larvae habitat associations likely reflect adult habitat preferences, spawning requirements and life history (Balon 1975), as well as mortality, which is extremely high for many species (Houde 1989; Partridge and DeVries 1999; Santucci and Wahl 2003). By targeting late-stage larvae and juveniles, my study was designed to acquire data that reflect active habitat selection within vegetated beds and provide information on which species continue to use aquatic vegetation as nursery habitat into later stages.

I also did not assess diurnal versus nocturnal use of vegetated habitats. Both Round Goby and Yellow Perch exhibit diel migrations (Post and McQueen1988; Hensler and Jude 2007), and previous research in other coastal wetlands indicates that species richness is generally higher at night, either due to diel migrations or gear selectivity (Midwood *et al.* 2016). While fish may exhibit migrations in the SLRE, the estuary's darkly stained waters and relatively high turbidity suppresses typical day-night differences in fish catch relative to other coastal wetlands (Hoffman et al, 2016), and likely subdues YOY diel migrations that are influenced by predation risk (Hansen and Beauchamp 2015).

Considering the substantial number of fish that depend on coastal wetlands (Trebitz and Hoffman 2015), there is a need for further microhabitat scale research to determine the functional role of aquatic vegetation and its effect on

larval and juvenile growth, mortality, and year-class strength. This topic requires considerable attention to facilitate both the preservation and restoration of this critical habitat. I found that aquatic vegetation plays a key role in providing nursery habitat within Great Lakes coastal wetlands, and that YOY fish assemblages vary along a depth and aquatic vegetation microhabitat gradient. To better inform coastal wetland restoration efforts, characterizing larval- and juvenile-habitat associations in wetlands dominated by plant species other than water celery is needed. In a broader sense, to better measure coastal wetland restoration success, it is paramount to consider how YOY fish assemblages respond to the structure and form of SAV in both restored and preserved coastal wetlands.

Chapter 2: Microhabitat associations of Tubenose Goby (*Proterorhinus semilunaris*)

Introduction

Non-native fish introductions are a leading threat to native fishes in North America (Jelks *et al.* 2008). In the Laurentian Great Lakes, non-native fish introductions have increased in conjunction with increasing shipping traffic, particularly during the period 1950 to 2000 (Mills *et al.* 1993; Ricciardi 2001). Several fishes, including the Round Goby (*Neogobius melanostomus*), Eurasian Ruffe (*Gymnocephalus*) and White Perch (*Morone americana*) are invasive, meaning they are widespread and either present a risk to native fish communities, disrupt food webs, or degrade habitats (French and Jude 2001; Gozlan et al 2010; Rush *et al.* 2012; Kapuscinski *et al.* 2015). Six invasive species (Eurasian Ruffe, Round Goby, Tubenose Goby [*Proterorhinus semilunaris*], White Perch [*Morone americana*], Common Carp [*Cyprinus carpio*], and Rudd (*Scardinius erythrophthalmus*]) occupy coastal wetland systems during most or all life stages (Table 1); however, coastal wetlands provide critical nursery habitat for many native fishes (Chubb and Liston 1986; Petering and Johnson 1991; Hook *et al.* 2001; Tanner *et al.* 2004; McDonald *et al.* 2014; Hoffman *et al.* 2015). Moreover, coastal wetlands are used by most Great Lakes fishes during some portion of their life history (Uzarski *et al.* 2005; Sierszen *et al.* 2012; Trebitz and Hoffman 2015). Thus, describing the niche of non-native fishes that occupy coastal wetlands is important for the conservation and management of Great Lakes fish communities.

The Round Goby and Tubenose Goby use coastal wetlands for spawning and nursery habitat. Native to the Black and Caspian Seas, both Round Goby and Tubenose Goby were first discovered in Lake St. Clair in 1990 (Jude *et al.* 1992). In just five years, Round Goby dispersed to all five Great Lakes, faster than any previous non-native fish (Charlebois *et al.* 2001; Kornis *et al.* 2011). Round Goby compete with several native forage fish species, particularly Mottled Sculpin (*Cottis bairdi*), Logperch (*Percina caprodes*), and Northern Madtom (*Noturus stigmosus)* for food resources (French and Jude 2001), and predate on Mottled Sculpin and Smallmouth Bass (*Micropterus dolomieu*) eggs (Janssen and Jude; Steinhart *et al.* 2004; Bergstrom and Mensinger 2009). Unlike native benthic fish species, the Round Goby feeds primarily on zebra mussels (*Dreissena polumorpha*) another Ponto-Caspian species that contributed to the rapid establishment and spread of the Round Goby throughout the Great Lakes (French and Jude 2001).

In contrast, Tubenose Goby have spread relatively slowly and have yet to spread to all five Great Lakes. They have been detected in the Lake St. Clair-Lake Erie corridor, western Lake Erie (Kocovsky *et al.* 2011), and more recently eastern Lake Erie (Grant *et al.* 2012) and far eastern Lake Ontario (NAS database). The St. Louis River in western Lake Superior also contains an isolated population (Dopazo *et al.* 2008; Peterson *et al.* 2011). Owing to their limited distribution, the ecological impacts associated with Tubenose Goby establishment are presumably less severe than those associated with Round Goby establishment

(Vanderploeg *et al.* 2002). However, there are few studies detailing the life history and habitat associations of Tubenose Goby, or their potential impact on native fish species within Great Lakes coastal wetlands (Jude and DeBoe 1996; French and Jude 2001; Dopazo *et al.* 2008).

Tubenose Goby are small (< 13 cm total length), benthic, nest-guarding fish found almost exclusively in shallow (<5 m), low velocity waters in both their native (Gaygusuz *et al.* 2010), and invaded systems (Jude *et al.* 1992; Peterson *et al.* 2011; Kocovsky *et al.* 2011; Janac *et al.* 2012). Within the Great Lakes, they are captured in or near aquatic vegetation beds (Jude *et al.* 1992; French and Jude 2001; Peterson *et al.* 2011). Many native species use aquatic vegetation as spawning substrate (Lane *et al.* 1996) and nursery habitat (Chubb and Liston 1986; Petering and Johnson 1991; Hook *et al.* 2001; Tanner *et al.* 2004; McDonald *et al.* 2014); thus, understanding the specific aquatic vegetation use and microhabitat associations of Tubenose Goby is the first step towards predicting their potential impact on native species (Kocovsky *et al.* 2011). Additionally, detailed Tubenose Goby habitat associations with water quality parameters, aquatic plant forms, and aquatic vegetation density are needed to make preliminary conjectures about their potential for range expansion. Recent expansions in Lake Erie may be facilitated by increases in aquatic vegetation (Kocovsky et al 2011), therefore more research is needed to identify how Tubenose Goby may use coastal wetlands that are undergoing both natural and human-induced habitat alteration. Many restoration efforts are aimed at fostering

aquatic vegetation growth (GLRI 2010), so elucidating these associations is also necessary to assess how restored wetlands may impact Tubenose Goby population abundance and expansion.

The objectives of this study were to document the distribution and abundance of young-of-year (YOY) Tubenose Goby within aquatic vegetation beds, to assess their microhabitat associations, and to describe the YOY fish assemblage to evaluate native species that share similar microhabitats. I sampled YOY fishes across a gradient of microhabitat conditions in the St. Louis River estuary (hereafter referred to as SLRE), a drowned river mouth coastal wetland system in western Lake Superior. In conjunction, I characterized both water quality, productivity and aquatic vegetation form and density to assess Tubenose Goby habitat associations at the microhabitat scale.

Methods

Study and Site Locations

The Port of Duluth, Minnesota-Superior, Wisconsin is the largest international shipping port (by cargo tonnage) in the Great Lakes and a non-native species introduction “hotspot” (Grigorovich *et al.* 2003). It is located within the drowned river mouth system formed where the St. Louis River meets Lake Superior. Upriver from the port, the SLRE contains numerous bays, tributaries, and islands creating a mosaic of SAV beds and emergent wetlands (Angradi *et al.* 2013). Allouez Bay, an isolated embayment behind a barrier beach to the east of the

port, also contains a variety of emergent and submerged aquatic vegetation (SAV, Tanner *et al.* 2004). The resident and migratory fish assemblage in the estuary is diverse (at least 60 species; J. Hoffman, unpublished data), including at least 10 non-native fishes (Peterson *et al.* 2011; Hoffman *et al.* 2016). Tubenose Goby were first detected in the river in 2001. Their establishment and population increase was poorly detected by bottom trawl surveys in the years since (Leino and Mensinger 2016), but by 2006, littoral sampling indicated they had become widespread and abundant in shallow habitats (Peterson *et al.* 2011).

I selected 12 vegetated sites spread across the SLRE (Fig. 1). They were selected to vary in both the density and diversity of aquatic vegetation types available within coastal wetlands. Three microhabitats were selected within each site based on a visual assessment of the abundance of different aquatic plant forms. These microhabitat categories included floating leaf plant abundant (n=13), emergent stands (n=10), and three SAV-only categories that were defined based on percent areal cover (estimated visually); low (<50%, n=2), medium (51-70%, n=5), and high (>70%, n=5; n=13 SAV only in total).

Fish collection

A single transect (25m) was designated within each microhabitat for fish sampling. Young-of-year (YOY) fish were collected using a benthic sled (1 m x 1 m net opening, length 3 m, 500 µm mesh net) that was pulled along this 25 m transect within each microhabitat. To ensure that the benthic sled did not become caught on logs and was properly deployed, fish were only collected during

daylight hours. Only YOY fish were retained; juvenile and adult fish were identified and released. Material in the net, including vegetation and debris, was triple-rinsed in river water, sieved (500 µm), and preserved in 95% ethanol. In the laboratory, each fish specimen was measured (total length [TL], ± 1 mm) and identified to species (Auer 1982), except *Lepomis* spp (sunfish); Bluegill (*Lepomis macrohirus)* and Pumpkinseed (*Lepomis gibbosus)*, which are both common in the river, can hybridize and are difficult to distinguish as larvae. Ten percent of fish from each size class were independently confirmed by a second taxonomist; no discrepancies were found.

Water quality measures and zooplankton collection

From 26 July through 3 August, 2016, temperature, dissolved oxygen, specific conductivity, pH and turbidity were measured *in situ* at the start and end points of each transect, 0.5 m below the surface (Hydrolab HL4 multiparameter sonde, OTT Hydromet, Loveland, CO). Also, two subsurface water samples (0.5 m depth; 1L) from the start and end of each transect were composited for chlorophyll-*a* analysis. Samples were placed in a cooler for transport, and vacuum filtered (47 mm GF/F) in the laboratory within 9 hours of collection. Chlorophyll-*a* concentrations were determined using fluorescence (Trilogy® Laboratory Fluorometer, Turner Designs, San Jose, CA) after a 24-hour 90% acetone extraction (EPA Method 445.0).

I collected a vertically-integrated sample (0.25 m off bottom to surface) of zooplankton using a diaphragm pump (Jabsco 3460). Samples from each end of

the transect (standard 40 L sample) were composited and sieved (500 µm). Zooplankton were preserved in 95% ethanol. In the laboratory, samples were further sieved (1000 µm) to remove algae. To ensure that sieving was not selecting specific taxa, both portions of five zooplankton samples (500-1000µm and >1000µm) were sorted by order and compared; for both size categories, cladocerans composed ~90% of zooplankters by abundance. The >1000µm samples were then filtered onto pre-weighed 500µm mesh filters, dried for 24 hours at 60°C, and weighed (AX2015 Delta Range balance; ±1 mg).

Aquatic Vegetation

In early August, I conducted a hydroacoustic survey of each site using a down-looking, single beam transducer (Biosonics DT-X digital echosounder; 6° beam, 420 kHz). Data were collected using Visual Acquisition software (version 6.2, Biosonics Inc., Seattle), transmitting 5 pings per second to obtain, on average, 4.5 pings per meter (linear distance). A 10 x 10 m coordinate grid was created in Arcmap and loaded onto an onboard GPS unit. This grid was used to establish hydroacoustic survey transects that were10 m apart and positioned perpendicular to fish sampling transects when possible.

In addition, between 30 August and 7 September 2016, I surveyed aquatic vegetation at each transect, using a stratified random design to assign sample locations. At each stratum: beginning, center, end of each transect, a randomly selected grid point (developed for the hydroacoustic survey) was selected for sampling. I placed a 1 x 1 m quadrat at each selected grid point, then assigned a

total percent cover estimate for the area within the quadrat. All plants were then removed (by hand or with a rake), identified to species, and assigned a percent cover value by species (MPCA 2014). Any additional species surrounding the boat, but not within the quadrat, were also noted and were used in plant species richness calculations. Any plants unidentified in the field were collected, kept cool, and transported back to the laboratory for identification.

Plant survey data from quadrat sampling was used to calculate plant species richness and abundance-weighted Coefficients of Conservatism (wC). The wC metric is an aquatic vegetation-based assessment of overall wetland condition based on the abundance and numerical rating (C scores) of each individual species within a site. The C-values represent an individual species' habitat fidelity and disturbance tolerance. I used regionally assigned C values for Minnesota (MPCA 2014) for my calculation. The estimated species cover from each survey was also used to calculate the relative cover of plant forms (floating leaf, emergent, and SAV) for each microhabitat.

I used Visual Habitat software (version 2.0, Biosonics Inc., Seattle, WA) to analyze the hydroacoustic data, calculating summary statistics for groupings of ten consecutive pings. For georeferencing, I assigned the latitude and longitude associated with each group's center. I exported the georeferenced summaries to ArcMap 10.4.1 (ESRI, Redland, CA) and calculated average aquatic vegetation cover (percent pings classified as plant), average depth, and total biovolume for each transect, only including ping groupings within 25 m of the transect. To index

aquatic vegetation patchiness, the average aquatic vegetation percent cover coefficient of variation was also calculated.

Tubenose Goby microhabitat analysis

To determine whether any microhabitat variables were significantly related to Tubenose Goby distribution and abundance, I fit 19 candidate generalized linear models (GLM) using maximum likelihood. I fit negative binomial GLMs using a log-link function. Negative binomial GLMs are commonly recognized as successful models for count data when compared to Poisson models, particularly when the error distribution is non-normal and overdispersed (Vaudor *et al.* 2011), which was the case with my catch data. I calculated Akaike Information Criterion values for small samples sizes (AICc). The AICc values estimate the quality of a model based on both goodness of fit and complexity. I compared each model's AICc to a null model's (intercept-only) AICc; more parsimonious models (lower AICc) indicated variable associations with Tubenose Goby abundance and distribution. The best models (lowest AICc) specify the microhabitat variables that had the strongest association. All modelling was performed in R (R 3.3.2 statistical software, R Core Team).

Fish assemblage analysis

I characterized YOY fish assemblage patterns and assessed Tubenose Goby-native species microhabitat overlap using ordination and classification methods. Together these techniques provide a robust tool for assessing patterns in

community data structure (Ludwig and Reynolds 1988). Non-metric dimensional scaling (NMDS) ordination using Bray-Curtis similarity was computed from square root transformed species abundance data using the *metaMDS* function in the *vegan* package (Oksanen *et al.* 2016) in R statistical software (v3.3.2, R Core Team). Three species - Smallmouth Bass, Round Goby, and Logperch (*Percina caprodes*) - were classified as rarely encountered (>1% relative abundance and >5% site occurrence) and not included in my NMDS and cluster analysis. Common Carp (*Cyprinus carpio*) was also removed owing to its skewed distribution (20 of 26 individuals were captured at the same microhabitat). Using the *hclust* function, I performed an UPGMA (Unweighted Pair Group Method with Arithmetic Means) hierarchical clustering analysis using the same Bray-Curtis similarity matrix as the NMDS ordination. Species with <1% relative abundance and <5% site occurrence were excluded from both analyses to avoid undue influence of rarely encountered species. I used a culled dendrogram (three largest groups) produced by clustering analysis overlaying my NMDS biplot to evaluate YOY fish assemblages and potential microhabitat overlap.

Results

I collected 527 individual Tubenose Goby ranging from 6 to 40mm total length (Fig. 5). Length data was normally distributed and indicated that the benthic sled effectively captured YOY Tubenose Goby, however did not capture adults (40-130mm). Tubenose Goby was the most abundant of the 12 taxa captured among all microhabitats sampled (Fig 6a), comprising 46.4% of the total catch. The

second most abundant taxon was sunfish. Tubenose Goby densities varied among microhabitats ranging from 0 to 3.96 fish/m^3 with an average density of 0.586 fish/m^3. Densities were generally higher in sites with aquatic vegetation cover over 64% (Fig. 6d) compared to sites with more exposure to wind and waves and less aquatic vegetation (Fig. 6b, 6c). The lowest densities occurred in the lower estuary and Allouez Bay (0 to 0.96 fish/m^3), which had less aquatic vegetation cover and a fish assemblage dominated by Black Crappie (Fig. 6b). Although not abundant at all sites Tubenose Goby were nearly ubiquitous, occurring within all 12 sites and at 75% of all microhabitats.

Tubenose Gobies were significantly associated with several microhabitat variables as indicated by GLM AICc values (Table 6). Several aquatic vegetation variables, including aquatic vegetation cover, patchiness, plant species richness and SAV cover performed better than the null (intercept only) model. Only two not aquatic vegetation variables, turbidity and zooplankton, had low AICc values. Aquatic vegetation cover was the best univariate model, however the best performing model overall, included both aquatic vegetation cover and plant species richness, indicating that Tubenose Goby are associated with aquatic vegetation structure and plant community character rather than solely cover. Together, the significant GLMs reveal that Tubenose Goby are most strongly associated with shallow, clear, densely vegetated and plant species rich habitats with abundant zooplankton.

Sunfish were the second most abundant taxa (316 individuals) with 72.2% microhabitat occurrence and dominated the assemblage in moderately vegetated habitats (Fig. 6c). Of all taxon captured, sunfish were most often captured in conjunction with Tubenose Goby. This relationship is also apparent in the NMDS analysis (Fig. 7).

Results from NMDS (stress = 0.18) and UPGMA cluster analyses indicated different YOY fish assemblages in the SLRE. The largest cluster or assemblage (Fig. 7, center), includes Tubenose Goby and indicates that they share habitat with several other species. They were most frequently captured in habitats with sunfish, Johnny Darter (*Etheostoma nigrum*), Tadpole Madtom (*Noturus gyrinus*), and Rock Bass (*Ambloplites rupestris*) indicated by their proximity in Fig. 7 and inclusion in the same cluster. Like the Tubenose Goby, both Tadpole Madtom and Johnny Darter are benthic species which presents higher potential for direct habitat overlap with Tubenose Goby.

UPGMA cluster analysis distinguishes a small group of microhabitats (Fig. 7 upper left quadrat) that contained very little aquatic vegetation cover. These microhabitats also contained no Tubenose Goby and relatively low fish densities (0.04 to 0.024 fish/m^3), indicating they may be overall poor habitat for YOY fishes. Another small cluster (Fig. 7 lower left quadrat) of sites from Allouez Bay depicts a Black Crappie dominated assemblage, where Tubenose Goby were sparse, reiterating the finding from GLM results that Tubenose Goby associate with clear water. Because Tubenose Goby were nearly ubiquitous in this study, it

is not surprising that they share habitat with several other wetland dependent species. However, taken in conjunction with GLM results, Tubenose Goby association with aquatic vegetation cover is strong, especially when compared to similar habitat association modelling for Rock Bass, sunfish, and Black Crappie (see Chapter 1 and Table 4).

Discussion

Although other successful coastal wetland invaders are tolerant of a broad range of environmental conditions and exhibit flexible habitat use (Mills *et al.* 1993; Trebitz *et al.* 2007; Pettit-Wade *et al.* 2015), I found the Tubenose Goby exhibited strong associations to specific microhabitat attributes, most notably aquatic vegetation cover and plant species richness. Tubenose Goby were abundant and ubiquitous within aquatic vegetation beds of the SLRE but were found in high abundance in clear water microhabitats with a rich plant community, abundant prey, and high aquatic vegetation cover (Fig. 8). These are characteristics commonly used to model fish habitat across Great Lakes coastal wetlands (Leslie and Timmons 1994; Brazner and Beals 1997; Croft and Chow-Fraser 2007). In Chapter 1, these same habitat characteristics indicated quality YOY fish habitat as indicated by fish species richness.

The specific habitat association of Tubenose Goby may explain their relatively slow expansion in the Great Lakes compared to the Round Goby. However, Tubenose Goby dominated assemblages in coastal wetland microhabitats with

high aquatic vegetation cover (Fig. 6d), and were strongly associated with high plant species richness. Thus, Tubenose Goby likely impact native species in these microhabitats, and may benefit from restoration efforts that convert previously unvegetated or sparsely vegetated habitats into dense, plant species rich, vegetated beds. Additionally, to detect recent Tubenose Goby introductions or expansion to new areas, fish surveys must target shallow vegetated habitats.

Our study indicates that YOY Tubenose Goby are strongly associated with a specific microhabitat within aquatic vegetation beds. This dependence on a specific microhabitat may contribute to their relatively slow expansion relative to the Round Goby, which displays high niche plasticity (Pettit-Wade 2015). Further, primary occupation of shallow, clear. and densely vegetated habitats likely limits Tubenose Goby dispersal around the Great Lakes via ballast water, which is taken from deeper habitats. Round Goby, which nocturnally forage in deep habitats are susceptible to ballast water uptake (Kornis *et al.* 2012). However, because this study is limited in its temporal scale, diel and seasonal use of these habitats is still unknown and requires further research to assess invasion risk. Nighttime drift of Tubenose Goby larvae was documented in the Danube River system in Austria (Zitek *et al.* 2004), and there is preliminary evidence the Tubenose Gobies undergo seasonal movements in western Lake Erie (Kocovsky *et al.* 2011). Further research on home range size and natural dispersal capacity at different life stages is needed to make an accurate assessment of invasion risk. Although unlikely to be transfer via ballast water, Tubenose Goby has

extended its range. Most notably, it has spread readily throughout the connecting channels of the Lake St. Clair-Lake Erie corridor, and more recently to western Lake Erie (Kocovsky *et al.* 2011).

The SLRE contains a higher density of Tubenose Goby (Peterson *et al.* 2011) compared to Lake St. Clair (Dopazo *et al.* 2008) and Lake Erie (Kocovsky *et al.* 2011) despite a more recent introduction. The relative abundance of aquatic vegetation, especially in the upper river portion of the SLRE, may contribute to the high Tubenose Goby densities. However, differences in sampling effort and gear preclude direct comparison of the two regions. Within the SLRE, deep-water early detection surveys using trawls missed a rapid Tubenose Goby population increase from 2001 to 2006 (Leino and Mensinger 2016), stressing the importance including littoral, structurally complex habitats to early detection surveys (Peterson *et al.* 2011; Hoffman *et al.* 2016). This study indicates that the benthic sled is effective at capturing YOY Tubenose Goby in these densely vegetated habitats. If Tubenose Goby are limited by available vegetated habitat within Lake St. Clair and western Lake Erie, recent increases in SAV density may improve conditions for their expansion. Both reductions in urban-runoff and Dreissenid mussel introductions have led to increased water clarity in the lower Detroit River, which in turn has fostered higher densities of wild celery (*Vallisneria americana* [Schloesser and Manny 2007]) that may provide suitable habitat for Tubenose Goby.

In the Lake St. Clair-Lake Erie corridor, Tubenose Goby hide within the small crevices of riprap, rocky substrate, and aquatic vegetation to avoid predation (Jude *et al.* 1995; Leslie *et al.* 2002; Jude and DeBoe 1996). Within the Danube basin, they have been found to prefer cobble and rocky substrate (Janac *et al.* 2012). Coarse substrates are rare within the SLRE (Angradi *et al.* 2013), and aquatic vegetation provides the only suitable cover, partially explaining their high abundance in these habitats. Other species also use aquatic vegetation as cover from predation during early life stages (Crowder and Cooper 1982; Trebitz *et al.* 1997); however, my study indicates that Tubenose Goby dominate the YOY assemblage in vegetated wetland habitat, suggesting they are particularly successful. No other abundant native wetland-dependent fishes in my study (i.e. Black Crappie, sunfish, and Rock Bass) had a strong microhabitat association with aquatic vegetation cover (see Chapter 1).

The lack of YOY fish microhabitat associations studies from before Tubenose Goby invasion prevents us from determining whether the Tubenose Goby has displaced native species in this densely-vegetated habitat or filled an open habitat niche. My findings suggest that two other benthic fishes, Tadpole Madtom and Johnny Darter, share habitats with the Tubenose Goby during late summer. Johnny Darter, like Tubenose Goby, spawn on the underside of logs and rocks (Jude *et al.* 1995) and may compete with Tubenose Goby for spawning substrate and habitats as larvae. Additionally, French and Jude (2001) documented significant diet overlap between small (35-74mm) Tubenose Goby and Rainbow

Darter (*Etheostoma caeruleum*), a close relative of Johnny Darter, because both primarily consumed dipterans. Further studies, including diet analysis and habitat overlap at other life stages of Tubenose Goby, madtoms and darters are needed to assess potential competition for food resources.

Coastal wetland habitats with high plant species richness and structural complexity foster an abundance and diversity of fishes (Chapter 1; Randall *et al.* 1996; Brazner and Beals 1997; Hook *et al.* 2001; Cvetkovic *et al.* 2010). Aquatic plant community composition and abundance reflects water quality (Lougheed *et al.* 2001; Trebitz *et al.* 2007; Croft and Chow-Fraser 2007). Aquatic plants also provide shelter and refuge from predation for larval and juveniles fishes, thereby creating important nursery habitat (Crowder and Cooper 1982; Trebitz *et al.* 1997). Thus, many efforts to restore coastal wetlands aim to improve native fish habitat by fostering a diverse plant community (Nichols 1999; Croft and Chow-Fraser 2007; MPCA 2014). Paradoxically, I found that Tubenose Gobies strongly associate with plant species-rich habitats. As such, they may directly benefit from coastal wetland restoration efforts that aim to establish dense, species-rich plant communities. The opportunity for colonization would be greatest in restoration projects in regions of the Great Lakes where Tubenose Goby have already been introduced. This indicates a substantial need for further research assessing the impact of Tubenose Goby on native species as well as their larger ecological role.

Tables

Table 1: All aquatic plant taxa identified from random stratified aquatic vegetation quadrat sampling within the St. Louis River estuary. C scores are from MPCA (2014).

Common Name	Binomial Name	C Score	Relative Cover	Transect Occurrence
Emergent				
common arrowhead	*Sagittaria latifolia*	3	0.98 %	2.8 %
sessile-fruited arrowhead	*Sagittaria rigida*	7	4.72 %	11.1 %
arrowhead unspecified	*Sagittaria spp.*		0.22 %	13.9 %
unbranched bur-reed	*Sparganium emersum*	8	0.89 %	2.8 %
softstem bullrush	*Schoenoplectus tabernae*	4	0.95 %	2.8 %
wildrice	*Zizania aquatica*	8	0.03 %	2.8 %
Floating Leaf				
yellow pond lily	*Nuphar variegata*	6	1.54 %	22.2 %
white water lily	*Nympheae odorata*	6	7.47 %	38.9 %
floating-leaf pondweed	*Potamogeton natans*	5	3.62 %	16.7 %
long-leaf pondweed	*Potamogeton nodosus*	6	0.96 %	16.7 %
floating-leaf bur-reed	*Sparganium fluctuans*	8	4.85 %	8.3 %
bur-reed unspecified	*Sparganium spp.*		1.17 %	8.3 %
lesser duckweed	*Lemna minor*	5	0.002 %	5.6 %
common duckweed	*Spirodela polyrhiza*	5	0.05 %	2.8 %
Submergent				
coontail	*Ceratophyllum demersum*	2	7.5 %	47.2 %
northern milfoil	*Myriophyllum sibiricum*	7	1.15 %	25 %
whorl-leaf milfoil	*Myriophyllum verticillatum*	8	0.05 %	2.8 %
milfoil unspecified	*Myrophyllum spp.*		0.01 %	5.6 %
common bladderwort	*Utricularia vulgaris*	5	0.41 %	2.8 %
water stargrass	*Heteranthera dubia*	6	0.61 %	33.3 %
flat-stem pondweed	*Potamogeton zosteriformis*	6	1.01 %	19.4 %
clasping-leaf pondweed	*Potamogeton richardsonii*	5	3.34 %	41.7 %
small pondweed	*Potamogeton pusillus pusillus*	7	0.03 %	2.8 %
water celery	*Vallisneria americana*	6	51.87 %	91.7 %
slender naiad	*Najas flexilis*	5	0.27 %	19.4 %
slender waterweed	*Elodea nuttallii*	7	0.01 %	2.8 %
common waterweed	*Elodea canadensis*	4	5.98 %	25 %
water marigold	*Bidens beckii*	8	0.18 %	2.8 %
stonewort unspecified	*Nitella spp.*		0.14 %	16.7 %

Table 1: Summary statistics of all aquatic vegetation metrics, including range, median, coefficient of variation (CV; both among and within sites).

Variable	Range	Median	Among Site CV %	Within Site CV %
Hydroacoustic surveys				
Aquatic vegetation Cover (%)	8 - 98	68.71	33	18
Patchiness (vegetation cover CV %)	7.9 - 186	42.7	63	36
Biovolume (m^3)	36 - 2241	1095.29	46	33
Quadrat surveys				
Weighted C (wC) score	4.11 - 6.45	5.98	4	7
Plant species richness	1 - 14	4	52	47
Floating leaf plants (% cover)	0 - 24	0.38	131	102
Emergent plants (% cover)	0 - 22	0	161	112
SAV (% cover)	0 - 53	6.13	74	57

Table 3: Larval and juvenile abundance, total density, relative abundance (of all YOY captured; bold type highlights the four most abundant species), and transect occurrence (percent of transects in which the species was encountered) for all fishes sampled by benthic sled in the St. Louis River estuary.

Common Name	Species Code	Scientific Name	Abundance		Total Density (fish/m^3)	Relative Abundance	Transect Occurrence
			Larvae	Juveniles			
Centrarchidae							
Rock Bass	**RB**	***Ambloplites rupestris***	**29**	**27**	**0.062**	**4.9 %**	**58.3 %**
Bluegill/Pumpkinseed	**BG**	***Lepomis spp.***	**291**	**25**	**0.351**	**27.8 %**	**72.2 %**
Smallmouth Bass	SB	*Micropterus dolomieu*	1	2	0.003	0.3 %	8.3 %
Black Crappie	**BC**	***Pomoxis nigromaculatus***	**53**	**65**	**0.131**	**10.4 %**	**58.3 %**
Cyprinidae							
Common Carp	CC	*Cyprinus carpio*	25	1	0.029	2.3 %	13.9 %
Golden Shiner	GS	*Notemigonus crysoleucas*	27	7	0.038	3 %	8.3 %
Spottail Shiner	SS	*Notropis hudsonius*	5	7	0.013	1.1 %	16.7 %
Gobiidae							
Round Goby	RG	*Neogobius melanostomus*	1	0	0.001	0.1 %	2.8 %
Tubenose Goby	**TG**	***Proterorhinus semilunaris***	**355**	**172**	**0.586**	**46.4 %**	**75 %**
Ictaluridae							
Tadpole Madtom	TM	*Noturus gyrinus*	27	5	0.036	2.8 %	41.7 %
Percidae							
Johnny Darter	JD	*Etheostoma nigrum*	4	4	0.009	0.7 %	11.1 %
Logperch	LP	*Percina caprodes*	0	2	0.002	0.2 %	5.6 %
Total			818	317	1.261		

Table 2: Akaike Information Criterion (AICc) values and model coefficients for univariate negative binomial Generalize Linear Models. Parameters in the first column (explanatory variables) were individually modelled with each response variable; fish species richness, Rock Bass abundance, sunfish abundance, and Black Crappie abundance. Low AICc values indicate good relative model performance (within columns). Bolded numbers indicate AICcs that are lower than the null model. Coefficients indicate the direction of the relationship between each model parameter and each response variable.

Model parameter	AICc (coefficient)			
	Species Richness	Rock Bass	Sunfish	Black Crappie
Null (Intercept only)	136.7	127.3	218.9	160.2
pH	137.4 (-1.8)	127.6 (-2.34)	221.2 (0.35)	161.0 (1.87)
Depth	138.7 (0.12)	129.4 (0.47)	**217.6 (1.46)**	162.5 (0.2)
Dissolved oxygen	136.6 (-0.28)	129.2 (-0.14)	221.1 (-0.08)	162.5 (0.05)
Specific conductivity	138.7 (0)	**122.3 (0.08)**	220.8 (0.02)	161.5 (0.05)
Temperature	**134.9 (0.3)**	129.2 (0.11)	**211.8 (0.57)**	161.9 (-0.14)
Turbidity	**130.0 (-0.05)**	**119.4 (-0.06)**	**215.6 (-0.05)**	**159.6 (0.03)**
Chlorophyll-a	**132.8 (0.05)**	129.4 (-0.01)	220.5 (0.03)	162.6 (0)
Zooplankton concentration	**132.8 (2.71)**	128.5 (1.21)	220.3 (1.27)	162.2 (0.76)
Weighted C (wC)	138.5 (0.21)	129.1 (0.41)	220.2 (0.46)	**159.7 (0.97)**
Plant species richness	**134.5 (0.13)**	129.3 (0.04)	219.6 (0.11)	162.3 (-0.04)
Aquatic vegetation cover (%)	**131.1 (0.03)**	127.6 (0.02)	221.0 (0.01)	162.3 (-0.01)
Patchiness	**134.7 (-1.44)**	**126.0 (-1.48)**	**217.8 (-1.5)**	162.2 (0.55)
Emergent plant (% cover)	138.5 (0.03)	128.8 (-0.08)	219.6 (-0.12)	161.9 (-0.07)
Floating leaf plants (% cover)	**134.4 (0.08)**	129.4 (0.02)	**217.0 (0.1)**	162.5 (0.01)
SAV (% cover)	**134.0 (0.04)**	129.6 (0.01)	**217.3 (0.05)**	162.5 (0)

Table 5: Attributes of six invasive fish that utilize Great Lake coastal wetlands; including distribution (S=Lake Superior, M=Lake Michigan, H=Lake Huron, E=Lake Erie, and O= Lake Ontario), age at maturity, relative fecundity (relative to other species included in table), spawning habitat, coastal wetland use, and known impact on native communities. [1] Mills et al.1993, [2] Gutch and Hoffman 2016, [3] Kornis et al. 2012 [4], Vanderploeg et al. 2002, [5] Valova et al. 2015, [6] Jude and DeBoe 1996, [7] Dopazo et al. 2008, [8] French and Jude 2001, [9] Bur 1986, [10] Boileau 1985, [11] Trebitz et al 2007, [12] Hawes and Parrish 2003, [13] Hurley and Christie 1997, [14] Schaeffer and Margraf 1987, [15] Swee and McCrimmon 1966, [16] Trebitz and Hoffman 2015, [17] Miller and Crowl 2005, [18] Tarkan 2006, [19] Kapuscinski et al. 2012a, [20] Kottelat and Freyhof 1972, [21] Kapuscinski et al. 2012b, [22] Burkhead and Williams 1991. [23] Peterson et al. 2011

	Eurasion Ruffe [2] *(Gymnocephalus cernuus)*	**Round Goby** [3] *(Neogobius melanostomus)*	**Tubenose Goby** *(Proterorhinus semilunaris)*	**White Perch** *(Morone americana)*	**Common Carp** *(cyprinus carpio)*	**Rudd** *(Scardinius erythrophthalmus)*
Distribution [1]	S, M and H: high abundance in St. Louis River estuary and Chequamegon Bay.	S, M, H, O, E; high abundance in Lake Erie-Huron corridor and St. Louis River estuary.	S, H, E: high abundance in Erie-Huron corridor and St. Louis River estuary [4].	All Great Lakes: high abundance in O and E.	All Great Lakes: high abundance in E.	E, O, and southern M
Age at Maturity	2-3	Males 3-4; Females 2-3	1-2	1-2 [9]	3-4	Males 1-2; Females 2-3 [18]
Relative Fecundity	**Medium:** extended spawning period and potential for multiple spawning events per season.	**Low:** multiple spawning events and high hatch rates.	**Low:** potential batch spawning [5].	**Medium** [9]	**High:** often spawns twice a year [15].	**Medium** [19]
Spawning Habitat	Shallow water (<3m); Eggs adhere to various substrates.	Shallow water. Males aggressively guard clusters of eggs adhered to the underside of rocks or logs.	Shallow water (<3m). Males guard clusters of eggs adhered to the underside of rocks or logs.	Spawn in shoals in shallow water (<1m), with no bottom type preference, eggs attach to substrate [10].	Spawn in very shallow water (<0.3m); eggs adhere to vegetation [15].	Spawn in dense vegetation where eggs adhere to plants [20].
Coastal Wetland Use	Spawning grounds and nursery. Overwintering grounds for some juveniles and adults. Prefer sandy/silty substrate with little or no vegetation.	Spawning grounds and nursery. Associated with rocky habitat and depth 2m [23]. Sometimes associated with SAV in drowned river mouths.	Spawning grounds, nursery and adult habitat. Juveniles and adults prefer shallow water (<5m) and dense aquatic vegetation [6,7].	Facultative use. Tolerant of disturbed and highly turbid habitats [11,12].	Spawning grounds and nursery [16]. Juveniles and adults prefer vegetated habitat.	Spawning grounds, nursery and adult habitat. Prefer vegetated habitat.

Table 3 continued…

	Eurasion Ruffe *(Gymnocephalus cernuus)*	**Round Goby** *(Neogobius melanostomus)*	**Tubenose Goby** *(Proterorhinus semilunaris)*	**White Perch** *(Morone americana)*	**Common Carp** *(cyprinus carpio)*	**Rudd** *(Scardinius erythrophthalmus)*
Impact	Diet overlaps with Emerald Shiner, Yellow Perch, and Trout-perch.	Diet and habitat overlaps with several species of Sculpin and Darter; feeds prolifically on invasive mussels; predates on native fish eggs.	Diet overlaps with Logperch and Rainbow Darter [8].	Competes with native Sunfishes [13], predates on native fish eggs [14].	Consumes aquatic vegetation, suspends sediment while foraging, modifying habitat and impacting plant and macroinvertebrate communities [17].	Consume aquatic vegetation, impacting density and composition of plant communities [21]; hybridizes with Golden Shiner [22].

*Table 6: Components of negative binomial, generalized linear models used to explain Tubenose Goby abundance. Bold typeface indicates models that performed better than the null model. * indicates significant variables in each model (p<0.05).*

Explanatory Variable(s)	AICc	Intercept Estimate	Slope Estimates	
			Variable 1	Variable 2
Null (Intercept only)	246.4	2.68		
Chlorophyll-a	248.7	2.79	-0.01	
Aquatic vegetation cover	**229.7**	**-2.19**	**0.06***	
Depth	245.8	4.62	-1.74	
Dissolved oxygen	246.8	4.5	-0.28	
Patchiness	**238.9**	**4.53**	**-4.58***	
Plant species richness	**231.5**	**0.71**	**0.25***	
Specific conductivity	248.1	-2.09	0.03	
Temperature	248.3	-1.84	0.18	
Turbidity	**242.3**	**3.4**	**-0.05***	
Weighted C score	248.2	5.37	-0.47	
Zooplankton	**240.6**	**1.51**	**5.32***	
pH	247.7	15.35	-1.67	
Floating leaf plant cover	247.8	2.40	0.07	
Emergent plant cover	246.5	2.39	0.15*	
SAV cover	**236.0**	**1.39**	**0.07***	
Aquatic vegetation cover + plant species richness	**228.7**	**-1.47**	**0.04***	**0.14***
Aquatic vegetation cover + patchiness	**231.8**	**-1.37**	**0.06***	**-0.91**
Aquatic vegetation + zooplankton	**231.8**	**-1.99**	**0.06***	**1.04**
Aquatic vegetation + turbidity	**232.2**	**-2.00**	**0.07***	**0**

Figures

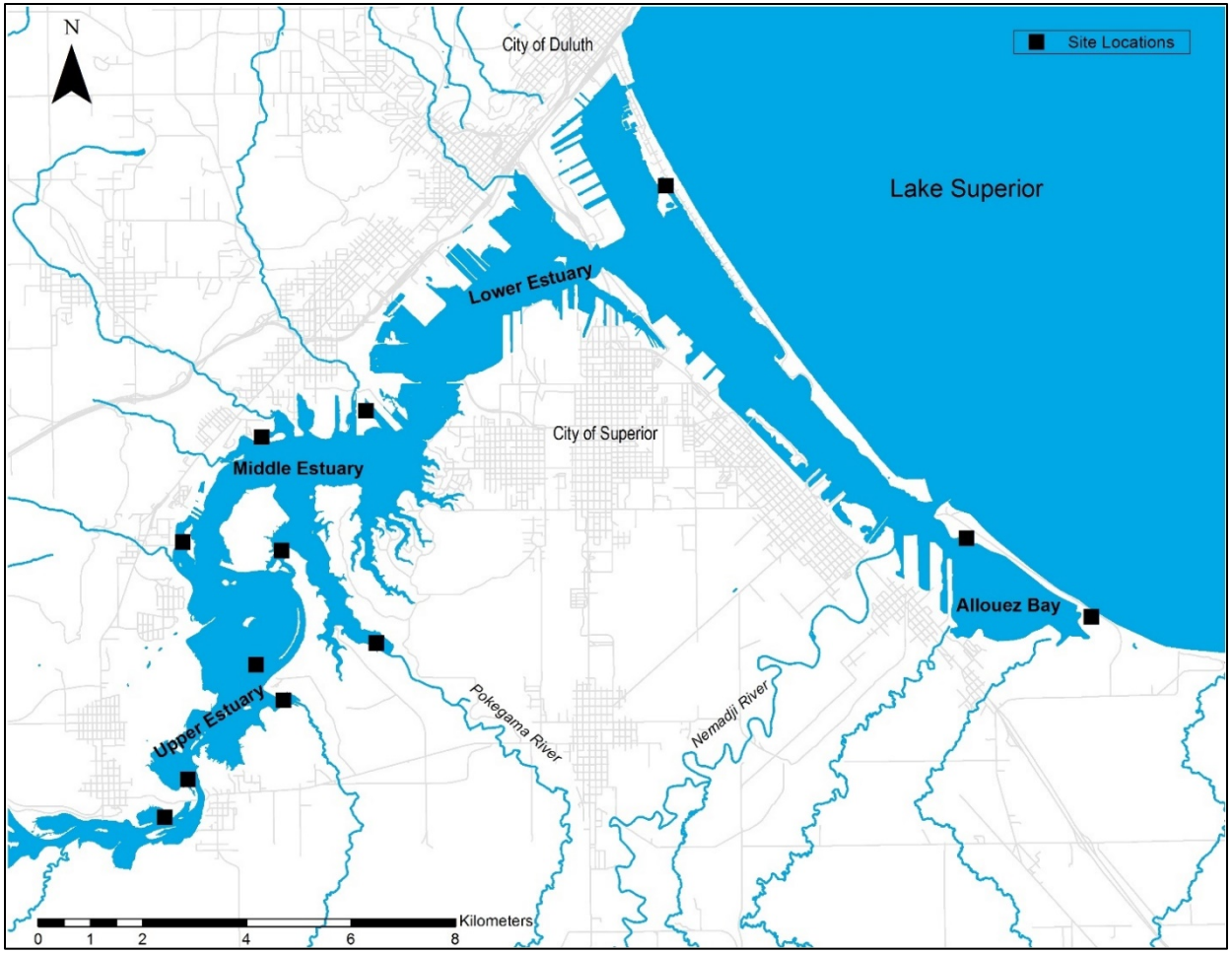

Figure 1: Map depicting the 12 vegetated beds sampled within the St. Louis River estuary from which young-of-year (YOY) fish and habitat data were collected during August 2016.

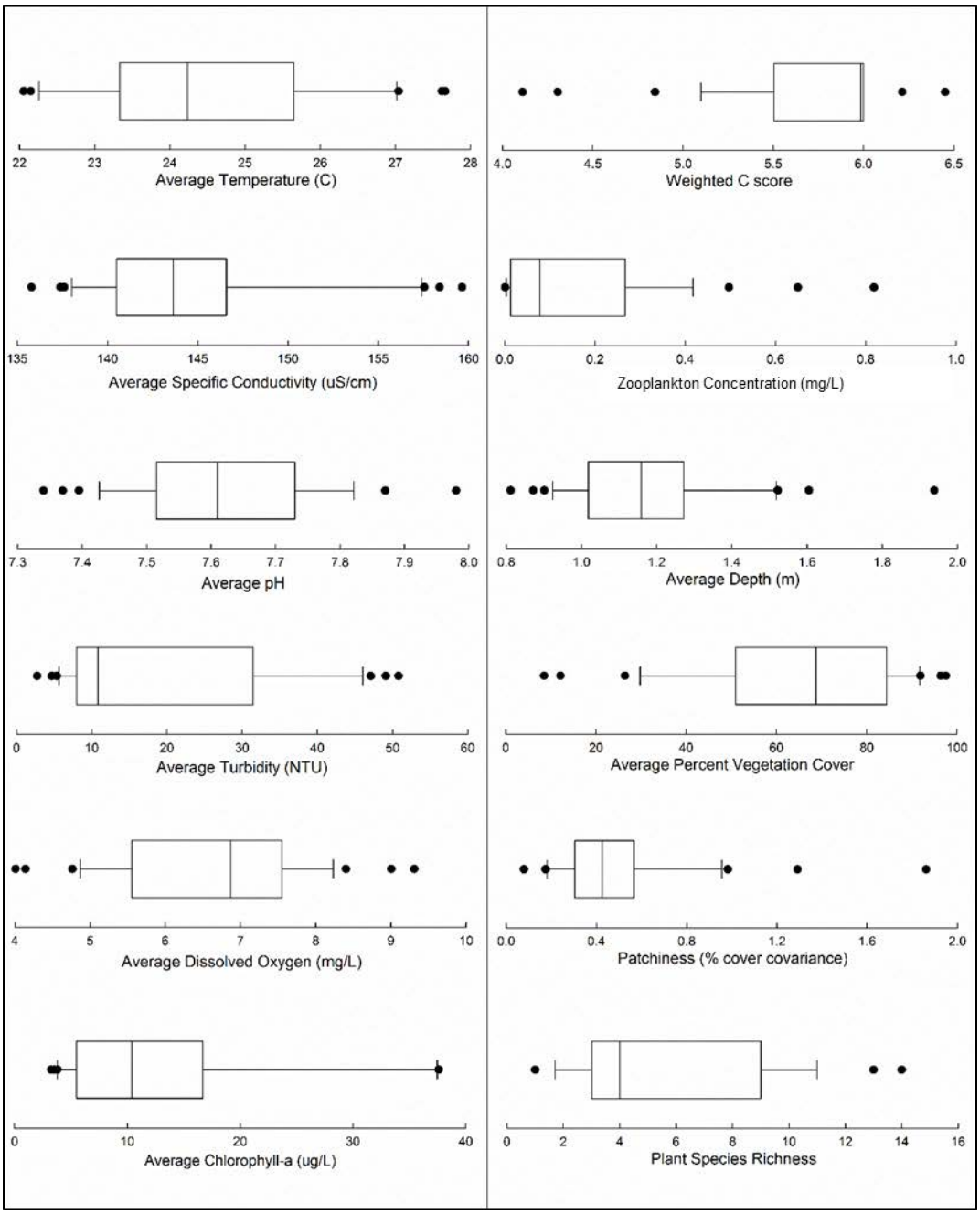

Figure 2: Box-and-whisker plots depicting the distribution of 12 environmental variables among transects (n = 36). The left and right edges of the box are the 1st and 3rd quartiles; the center line is the median. The left and right whiskers are the 10th and 90th percentiles of the data. The dots are outliers. Averages are based on within-transect replicates.

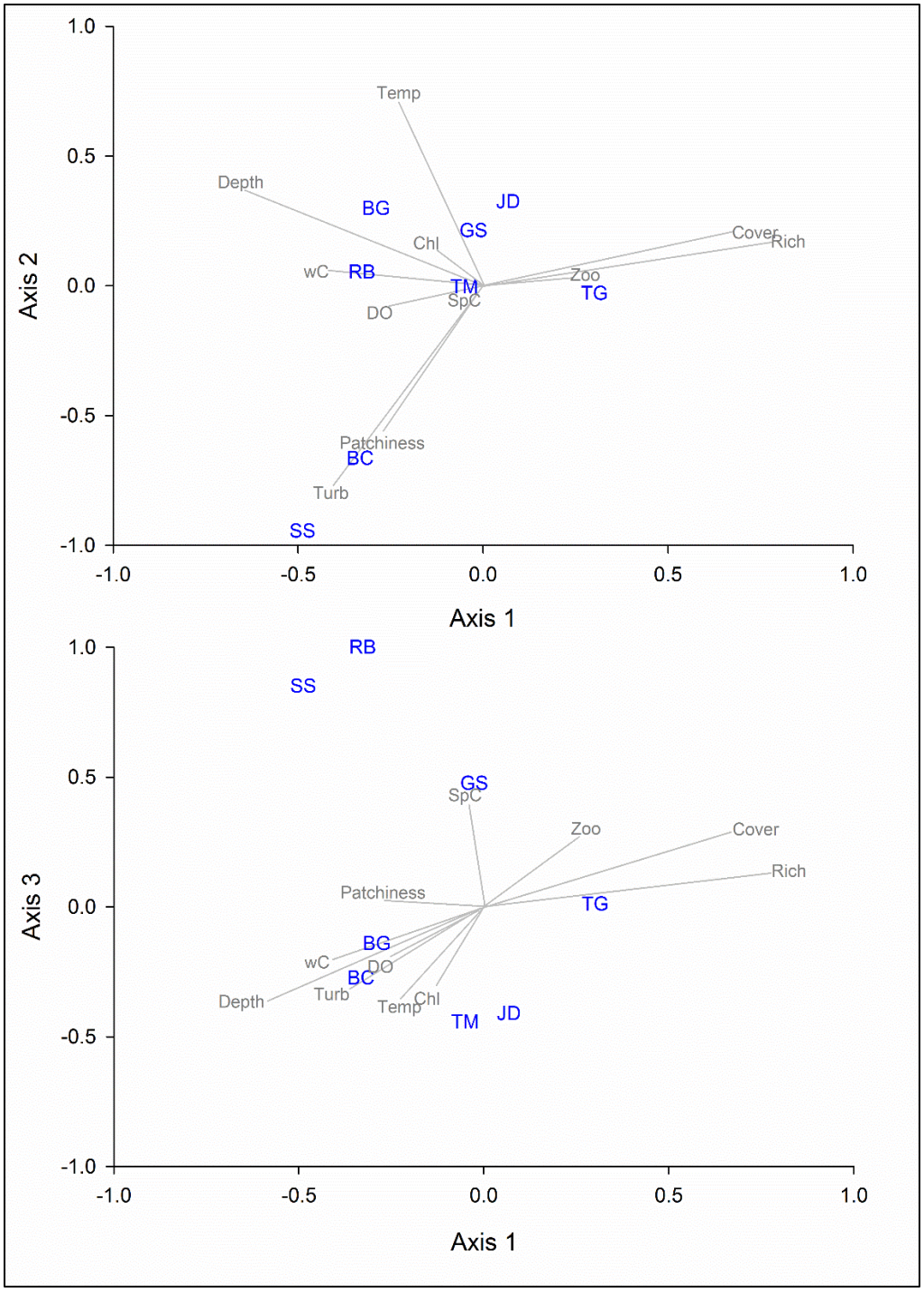

Figure 3: Canonical Correspondence Analysis (CCA) bi-plots illustrating the association of young-of-year fish species abundance to 11 environmental variables in the St. Louis River estuary. Each species is represented by a two-letter code (see Table 1). Environmental variables include specific conductivity (SpC), zooplankton concentration (Zoo), aquatic vegetation cover (Cover), aquatic plant species richness (Rich), chlorophyll-a concentration (Chl), temperature (Temp), dissolved oxygen (DO), turbidity (Turb), weighted C score (wC), depth (Depth), and patchiness (Patchiness). Species scores were rescaled (score/2) for presentation.

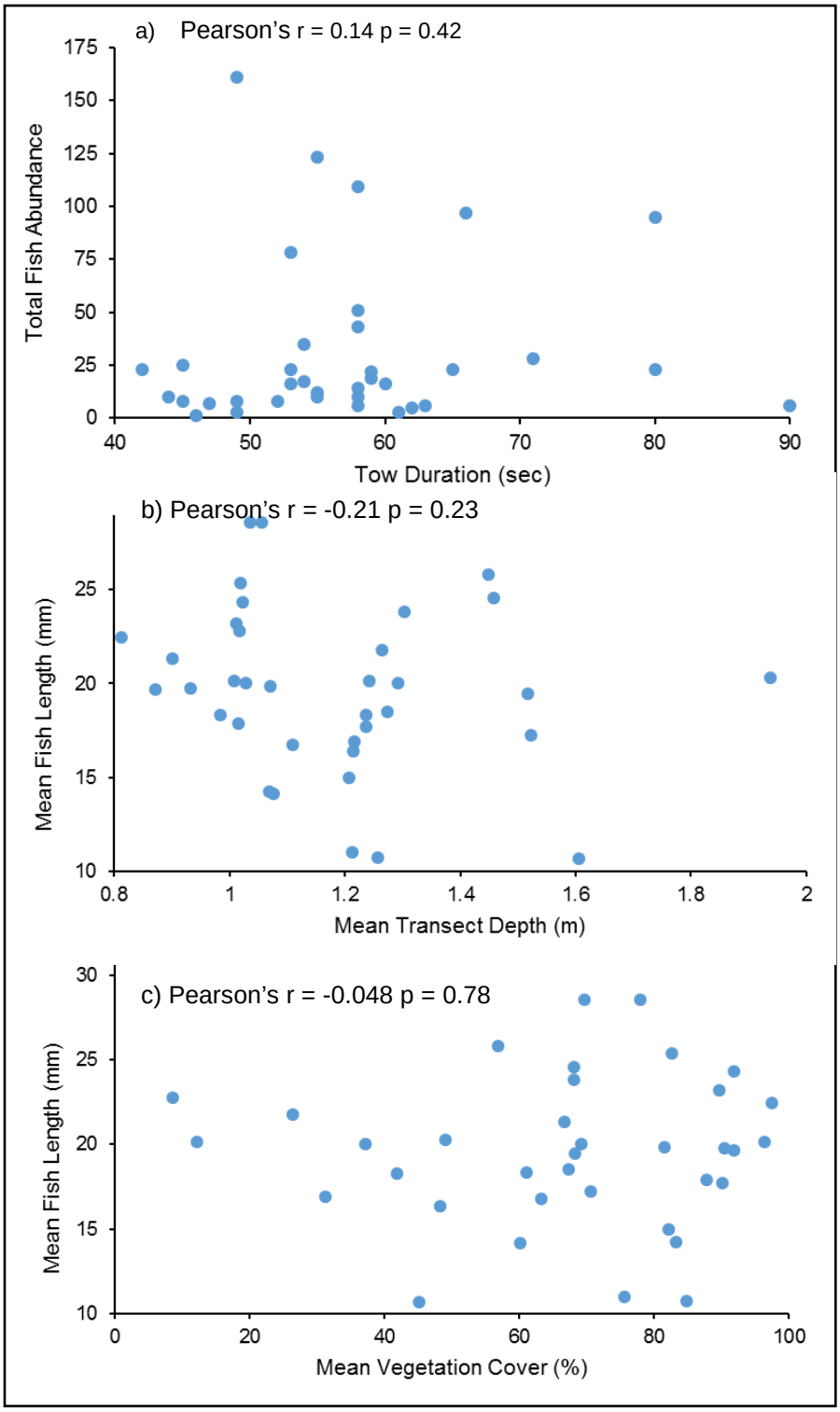

Figure 4: Total fish abundance with respect to tow duration (a), mean fish length (all species combined) with espect to mean transect depth (b) or mean transect aquatic vegetation cover (c), with estimated Pearson's r and p values.

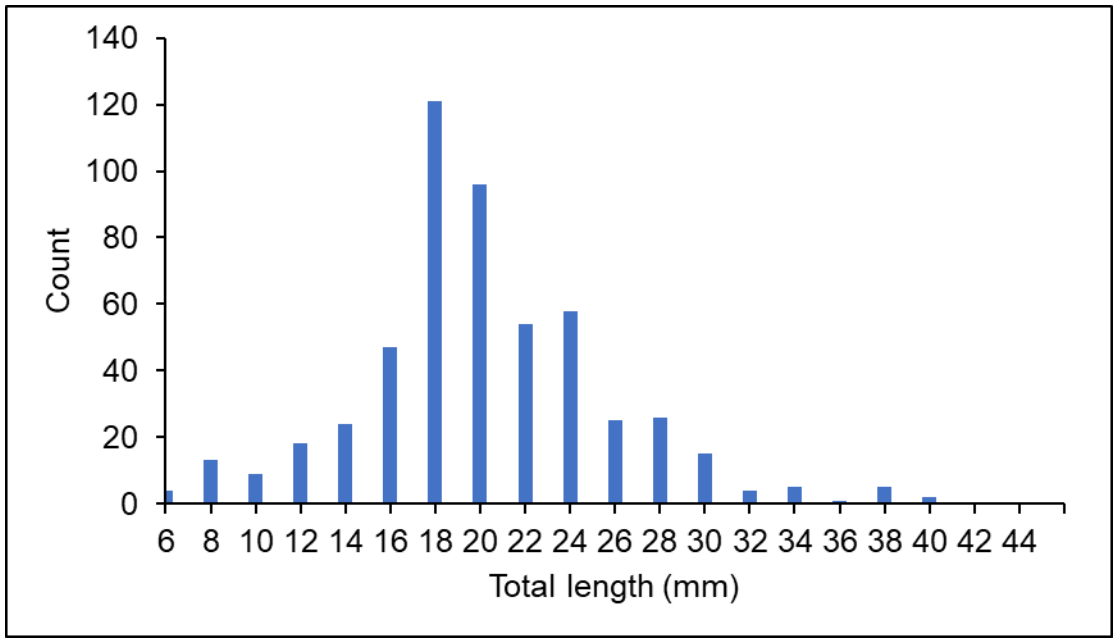

Figure 5: Length histogram of all Tubenose Goby captured among all 36 microhabitats. Mean = 19.7, median = 18.5, and standard deviation = 5.58)

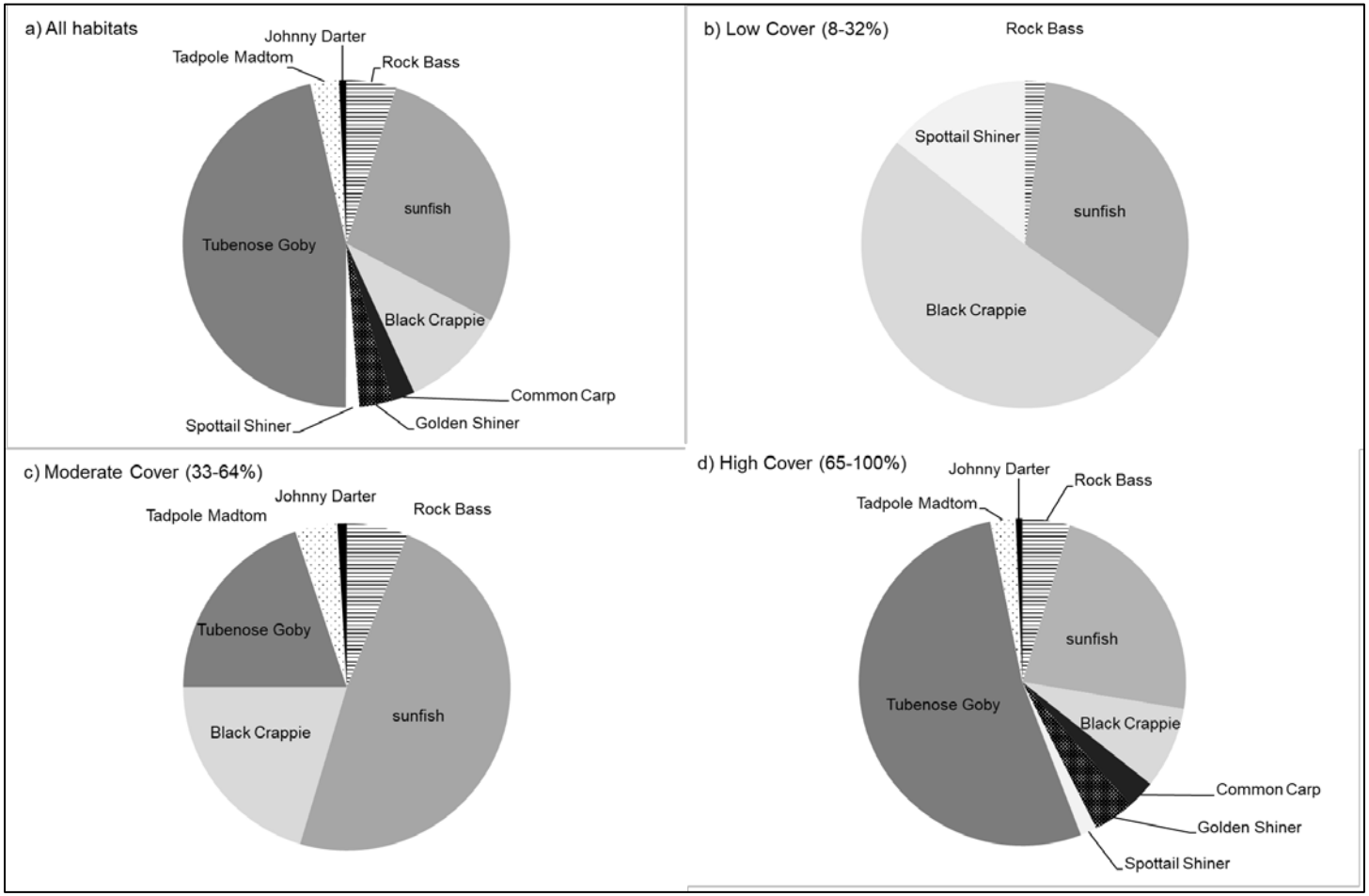

Figure 6: Relative abundance of fish species captured across the St. Louis River estuary in all 36 habitats sampled (a), and among habitats of low (b), moderate (c), and high (d) aquatic vegetation cover (based on hydroacoustic data).

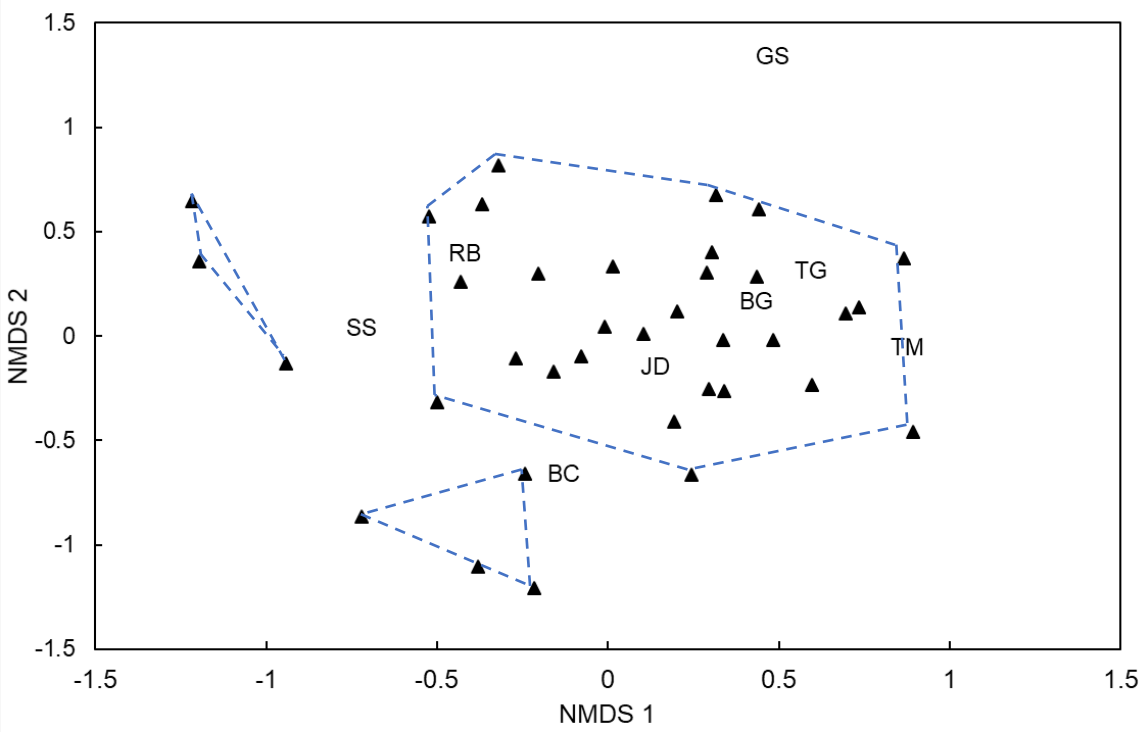

Figure 7: Patterns in fish composition across 36 microhabitats (triangles) displayed in a two-dimensional nonmetric multidimensional scaling (NMDS) ordination of species abundances (stress = 0.18) overlaid with the three largest clusters resulting from UPGMA clustering analysis (dashed lines). RB = Rock Bass, SS = Spottail Shiner, BC = Black Crappie, JD = Johnny Darter, BG = sunfish, TG = Tubenose Goby, GS = Golden Shiner, TM = Tadpole Madtom.

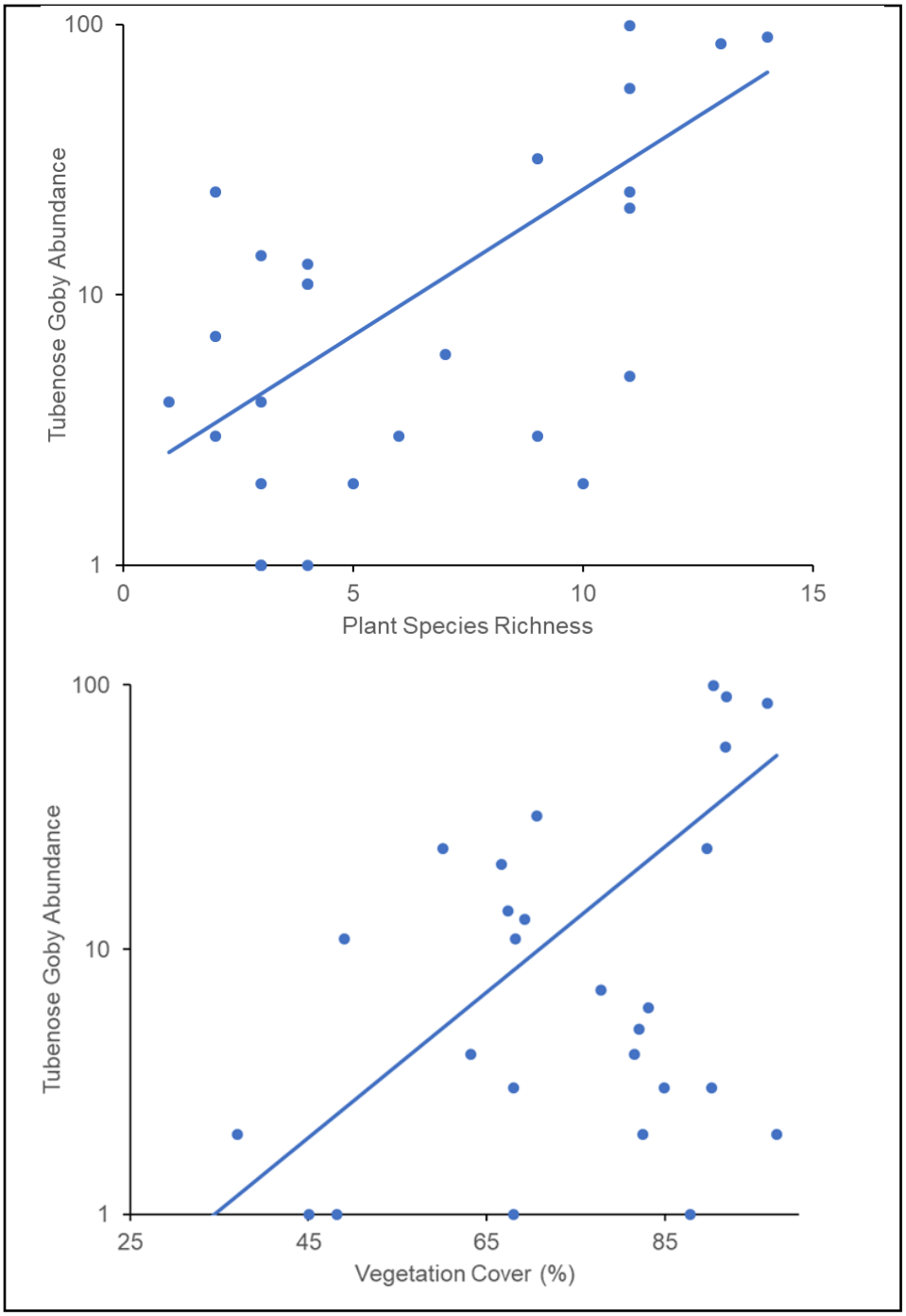

Figure 8: The relationship between Tubenose Goby abundance (number of tubenose goby per microhabitat) and both plant species richness (number of species per microhabitat) (a) and aquatic vegetation cover (b). Straight lines are the linear predictors of generalized linear models (parameters in Table 2).

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